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INVERTEBRATES AND RUNOFF IN A SMALL URBAN STREAM IN EDMONTON, ALBERTA

by

© ERIC ROSS WHITING

A THESIS

SUBMITTED TO THE FACULTY OF GRADUATE STUDIES AND RESEARCH

IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE

OF MASTER OF SCIENCE

DEPARTMENT OF ZOOLOGY

EDMONTON, ALBERTA

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THE UNIVERSITY OF ALBERTA

FACULTY OF GRADUATE STUDIES AND RESEARCH

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research, for acceptance, a thesis entitled "Invertebrates and Runoff in a Small Urban Stream in Edmonton, Alberta" submitted by Eric Ross Whiting in partial fulfillment of the requirements for the degree of Master of Science.

ABSTRACT

Invertebrates of Whitemud Creek, a small urban stream in the city of Edmonton, Alberta, Canada, were studied from April 1976 to May 1977. Emphasis was on assessing the effects of storm sewer discharge, which enters the creek at several points during and after rains and spring snowmelt.

Within the city, especially immediately below storm drain outlets, there were large decreases in the diversity of the fauna. These decreases were due both to the disappearance of many types of typically pollution-intolerant invertebrates and to large increases in the numbers of a few pollution-tolerant forms, especially tubificid worms and orthoclad midges. The sizes and growth rate of nymphs of the mayfly *Caenis forcipata* were higher at sites just downstream from storm drain outlets. These changes were apparently in response to nutrient enrichment, associated oxygen deficits, and increased silt deposition.

The magnitude of the faunal changes can be related to the area of the urban watershed. A model illustrating this relationship is presented. The construction of the Whitemud Freeway Bridge had little obvious effect, although bridge construction disturbance might have been masked by the effects of storm sewer discharge.

Seasonal variations in the invertebrate fauna are apparently affected by both runoff and life cycles. Minimum faunal diversity and richness occurred during spring runoff. There were also general decreases in diversity and richness throughout the study. I conclude that the stream's biota is more sensitive to storm sewer input than the chemical and physical constituents measured.

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INTRODUCTION

In recent years urban runoff has been receiving attention as a source of pollution in receiving waters. This is due in part to better control of domestic and industrial wastes and in part to the large increases of urban populations (Field 1975, Field and Knowles 1975, Field and Szeeley 1974, McElroy et al. 1975, Wanielista, Yousef and McLellon 1977). Several studies (Weibel, Anderson and Woodward 1964, Bryan 1972, Sartor, Boyd and Agardy 1974, Whipple, Hunter and Yu 1974, Wanielista et al. 1977) have shown that, in terms of total weight of material dumped annually into receiving waters, the contribution of urban runoff is roughly equivalent to that of secondarily treated domestic sewage. The input of total solids may be 10 to 20 times as great from urban runoff as from domestic sewage. The majority of these inputs probably enters receiving waters via storm sewer discharge, although little is known about the importance of direct overland flow.

Urban runoff carries such a large amount of material because of the quantity of material accumulating on urban surfaces and because of altered urban flow patterns. The large percentage of flat, non-porous surfaces in urban environments results in increased amounts and rates of runoff. Inputs of material in urban runoff are usually sporadic, occurring only during rains or spring thaws. Contributions of runoff flow at other times are negligible (Field and Knowles 1975).

The water chemistry of urban runoff has been widely studied. Stormwater runoff has been reported to contain nitrogen, phosphorus,

pesticides (chlorinated hydrocarbons and polychlorinated biphenyls), lead and other heavy metals, oil, salt, chromates, cyanide, organic materials, and suspended and dissolved solids (Field and Weigel 1973, Field 1975, Wanielista et al. 1977, Weibel et al. 1966). However little work has been done on the effects of urban runoff on the biota of receiving waters. Thus, my study was undertaken to examine the fauna of Whitemud Creek within the city of Edmonton, Alberta, and to determine the nature of the fauna's response to urban runoff.

DESCRIPTION OF STUDY AREA

Whitemud Creek is a small third-order stream located in central Alberta (53° 20' N, 113° 30' W). The stream has two main branches, Whitemud Creek proper, arising from a number of intermittent ponds southwest of Leduc, Alberta, and Blackmud Creek, arising from Saunder's Lake, east of Leduc. The confluence of the two streams takes place 10.5 km upstream from where Whitemud Creek empties into the North Saskatchewan River (within the city of Edmonton). The stream drains approximately 1030 km^2 of predominantly agricultural land.

Whitemud Creek basin is underlain by Upper Cretaceous, Edmonton Formation deposits of bentonitic sandstones, shales, clay and coal. Superficially the area is a low-relief till plain covered mainly with lacustrine deposits associated with glacial Lake Edmonton (Rains 1969). The soil is classified by Bowser et al. (1962) as chernozemic, and the topography is level to gently undulating.

The climate is continental with mean summer and winter temperatures of 13° C and -9° C, respectively. Average annual precipitation is 450 mm, of which 73% falls as rain and 27% as snow (Erxelben 1972).

Bowser et al. (1962) describe the vegetation as aspen parkland transitional to mixed boreal forest, with grasslands and aspen bluffs on the drier sites and balsam poplar and white spruce in the wetter areas, e.g. the stream valley. Most of the watershed upstream from the city is now used for agriculture, with cereal and forage cropland and pasture.

The study area encompasses that portion of Whitemud Creek lying within Edmonton's city limits. This is from the confluence of Whitemud and Blackmud Creeks just south of 23 Avenue to the mouth on the North Saskatchewan River. The stream channel in this area is 10.5 km (Fig. 1) and the stream drains 19.4 km^2 of multiple usage land. Fourteen percent of the area is natural woody vegetation, 34% is agricultural, 8% is parkland, 27% is residential, 7% is paved and 10% is disturbed by surface scarification (Fig. 2).

The stream valley has been left in more or less its natural condition, with a cover of aspen, balsam poplar, white spruce and numerous shrubs providing a buffer zone for the stream. Land to the west, south of 45 Avenue, has not yet been developed and is used for cereal grain production. North of 45 Avenue, there is an area of pasture, operated by the University of Alberta Experimental Farm. The areas classified as parks are areas having modified vegetation cover, such as golf courses. The residential areas consist primarily of single family dwellings and low-rise apartments, with some light commercial use. The two major disturbed areas are the new housing development on the east side of the stream near the city's edge and the site of the construction of the Whitemud Freeway Bridge. Both sites contain large areas of surface scarification.

In addition to the natural watershed described above, six separate storm drainage systems empty into Whitemud Creek (Fig. 3) (Plate 1). They are composed primarily of residential lots, low-rise apartments, city streets, and a few high-rise apartments. The commercial developments within the areas are all fairly small operations, with the exception of Southgate Shopping Center in storm sewer system II.

Plate 1. Storm sewer II outlet, typical winter flow, 1977.

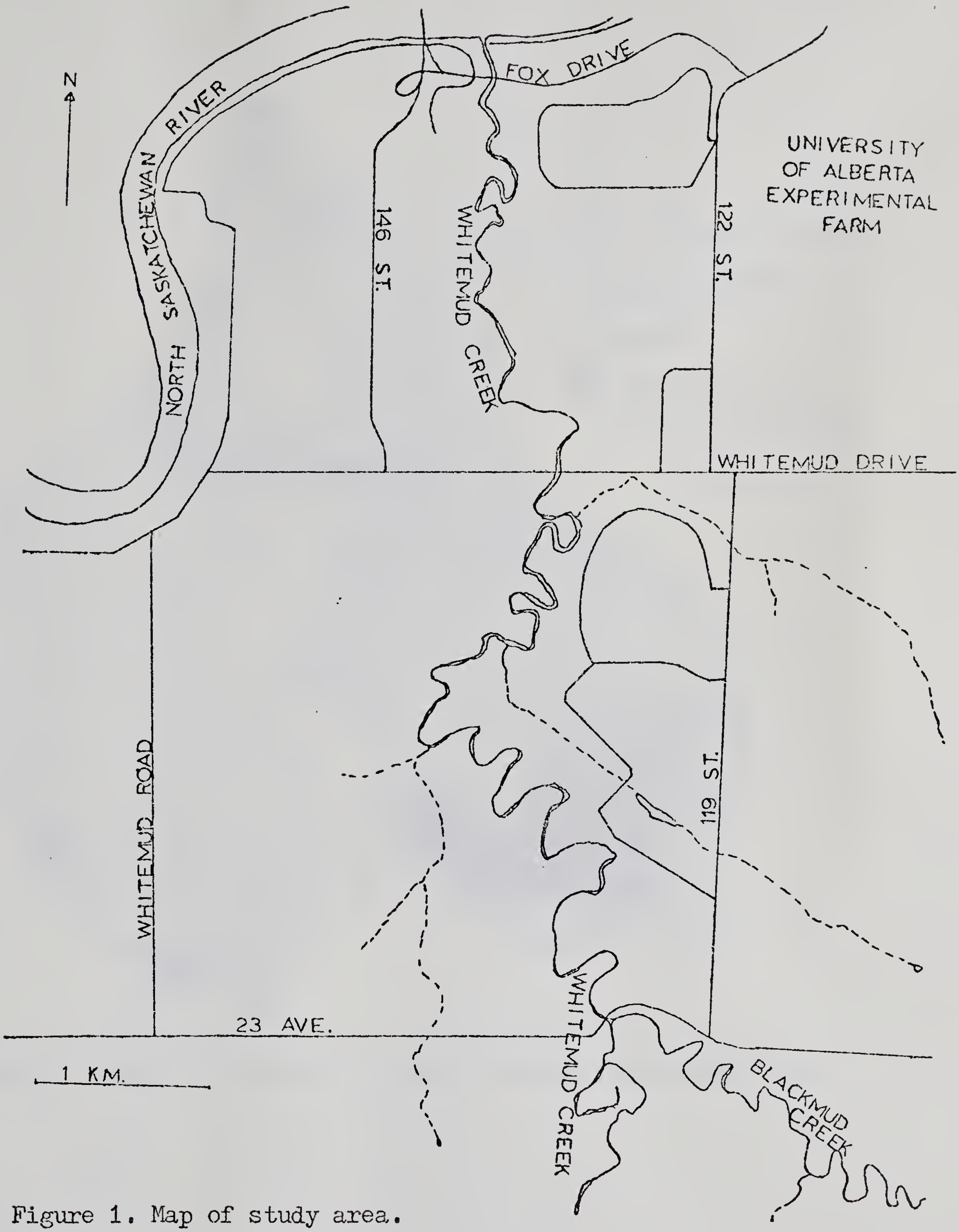


Figure 1. Map of study area.

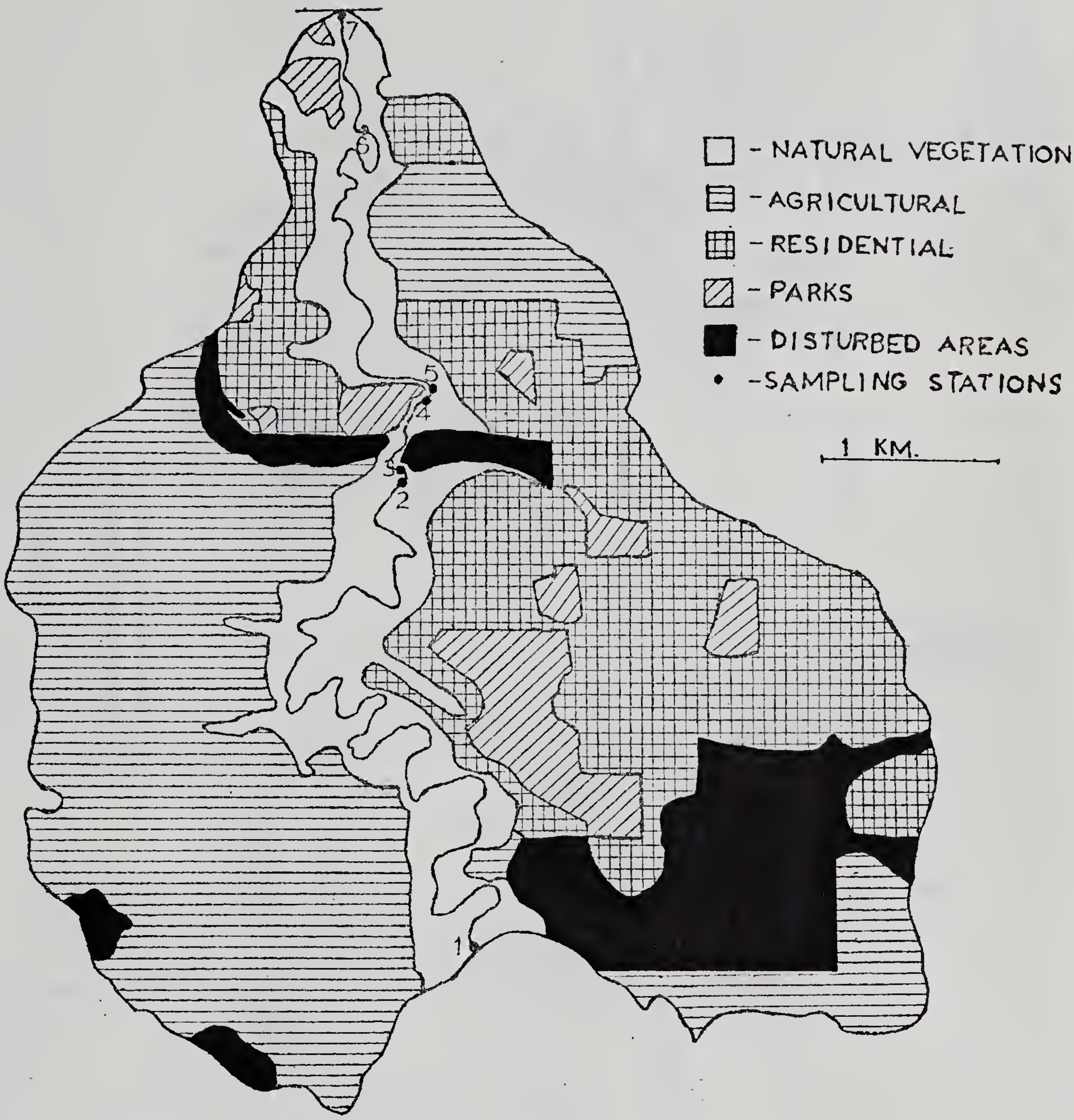
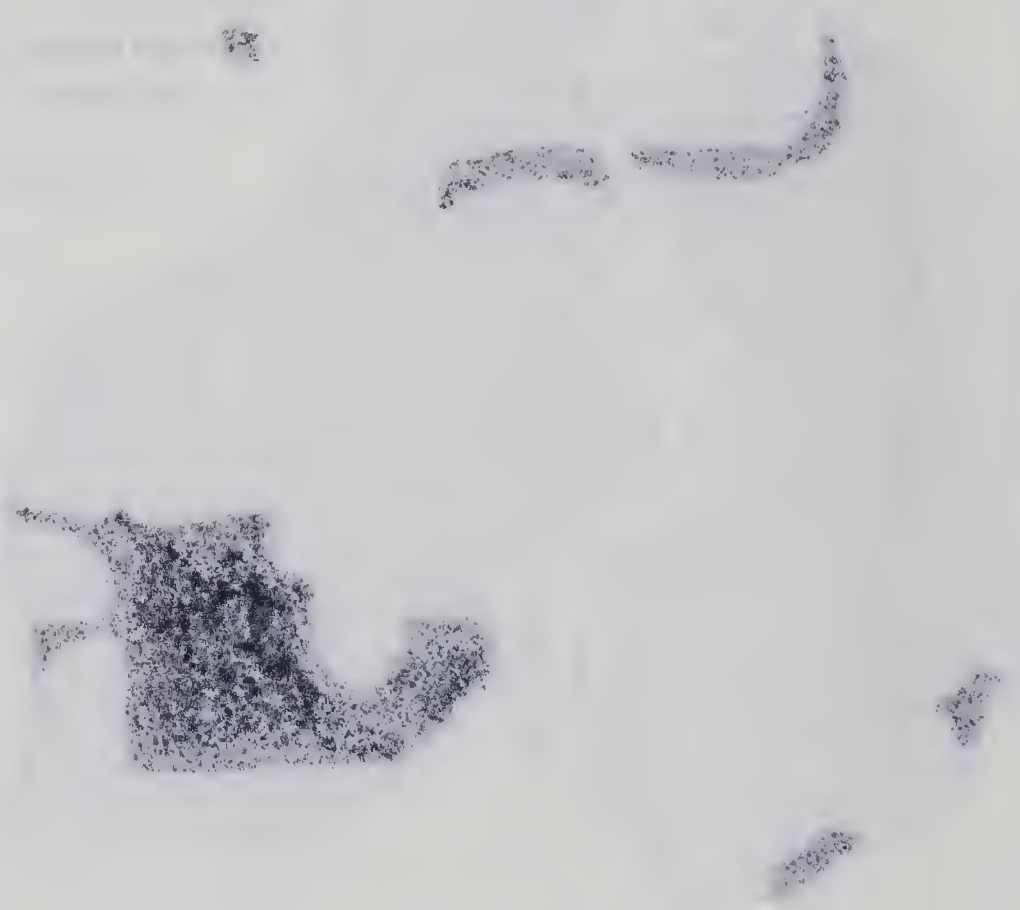


Figure 2. Map of study area showing land use in drainage basin.



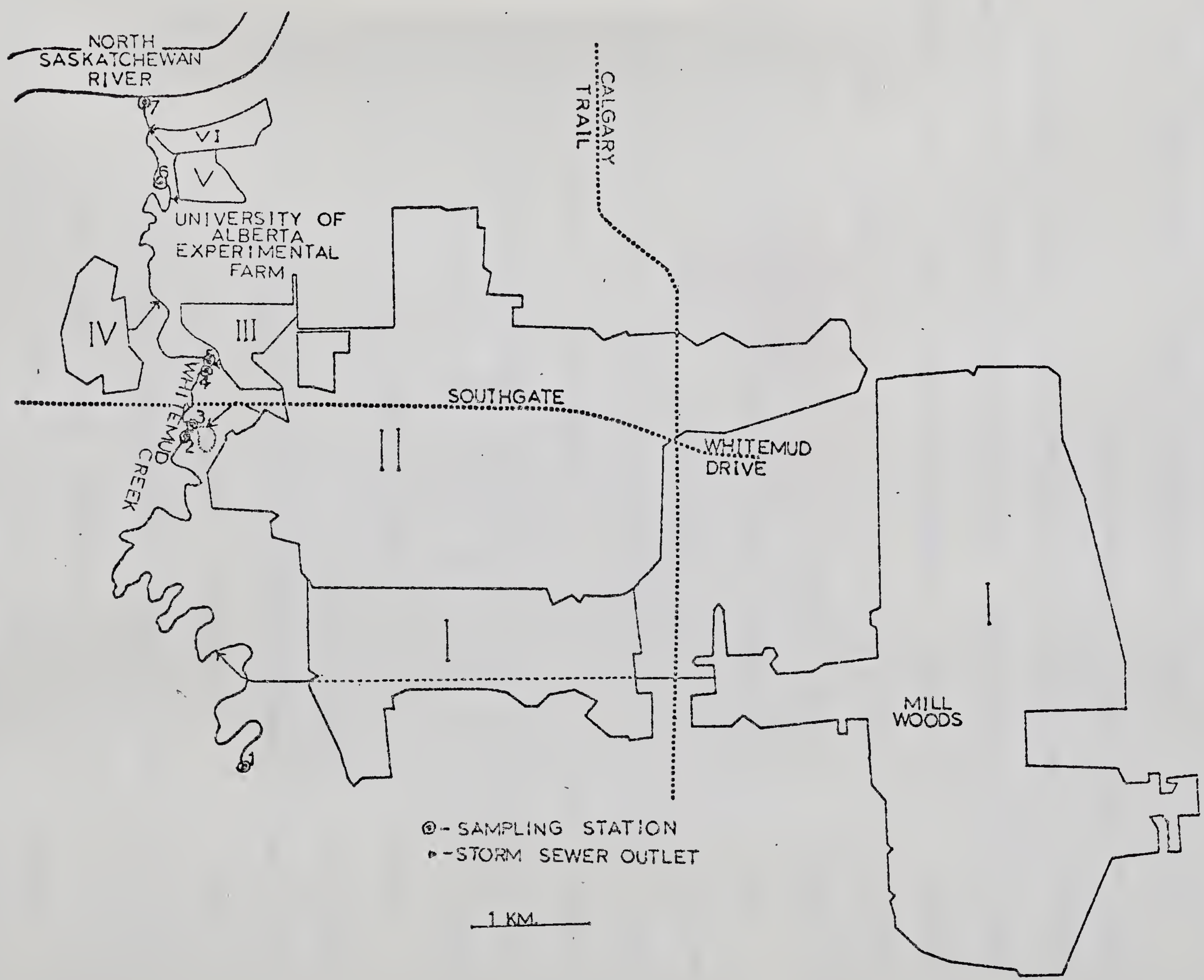


Figure 3. Map of study area showing storm sewer drainage areas.

The shopping center is surrounded by a large paved parking lot. The only industrial developments are some light industrial areas at the extreme east ends of storm sewer systems I and II.

Within the city, Whitemud Creek consists of alternating riffles and pools. According to the Cummins (1962) classification, the riffles are composed mainly of particles in the gravel and pebble fractions, with a few cobbles. Assorted sands and silts are deposited on and removed from the riffles during the annual flow cycle. The pools have a sapropel bottom.

Seven sampling stations were selected along Whitemud Creek within the city, all in riffle areas (Figs. 2 and 3). Site 1 is located just inside the city boundary, between the confluence with Blackmud Creek and the 23 Avenue bridge, 10.5 km upstream from the mouth. This site is upstream from any urban development in Edmonton and was chosen as a control site (Plates 2 and 3).

Site 2 is located 50 m upstream from the outlet of storm drain II, and site 3 is 10 m downstream from this outlet. This area is about 4.2 km upstream from the mouth and 0.4 km upstream from the Whitemud Freeway Bridge. Commencing in late June 1976 and continuing until the end of the study, there were massive surface disturbances along both banks of the stream between sites 3 and 4, associated with the construction of the new bridge.

Site 4 is 50 m upstream from the outlet of storm drain III, and site 5 is 5 m downstream from this outlet. This area is about 3.5 km upstream from the mouth and 0.3 km downstream from the Whitemud Freeway Bridge (Plates 4 and 5).

Plate 2. Site 1, April, 1977.

Plate 3. Site 1, August, 1977.

Plate 4. Site 5, April, 1977.

Plate 5. Site 5, August, 1977. Notice the *Cladophora* in the bottom left hand corner.

Site 6 is about 200 m downstream from the outlet of storm drain V, 0.8 km upstream from the mouth. This site was altered by the establishment of an outdoor skating rink on the stream during the winter of 1976-1977. The rink, in operation from November through March, resulted in the formation of a very thick layer of ice. There was no perceptible current at the study site during this period.

Site 7 is located just upstream from where Whitemud Creek empties into the North Saskatchewan River. The water level at this site fluctuated greatly, being dependent on the water level in the North Saskatchewan River. When the river level was high, Whitemud Creek backed up at this site. Although the current velocity was greatly reduced at these times, the integrity of the gravel bottom was maintained.

Six species of fish, longnose sucker (*Catostomus catostomus*), white sucker (*Catostomus commersoni*), northern chub (*Couesius plumbeus*), fathead minnow (*Pimephales promelas*), trout-perch (*Percopsis omiscomaycus*) and brook stickleback (*Culaea inconstans*), are found in Whitemud Creek throughout the year (Visscher 1977). Several additional species, including northern pike (*Esox lucius*), mountain whitefish (*Prosopium williamsoni*), longnose dace (*Rhinichthys cataractae*) and burbot (*Lota lota*), have been found in Whitemud Creek during the summer (J. Reist pers. comm.). The fish fauna of Whitemud Creek is dominated by white sucker, which accounted for more than half of the numbers collected by Visscher (1977) during the winter.

METHODS

Samples were collected from all seven sites at approximate 3 week intervals from April 1976 to May 1977. One or two sites were sampled per day, with all seven sites usually being completed within 4 to 7 days. Sites 6 and 7 were not sampled at the beginning of November 1976 because of ice conditions. During the time of greatest snowmelt, in February and March, sampling was conducted at 2 week intervals.

Physical-Chemical

Flow was measured using the cork float method (Welch 1948). Air and water temperatures were measured with an ASTM thermometer. Daily maximum and minimum air temperatures and precipitation were obtained from the Environment Canada Monthly Meteorological Summaries (1976, 1977) for the Edmonton International Airport. The percent cover of filamentous algal growth was estimated qualitatively.

Two replicate samples for dissolved oxygen analysis were collected from the middle of the riffle. They were fixed in the field according to the azide modification of the Winkler method, and brought to the laboratory where they were titrated within 24 hours.

Two one-liter plastic jars were filled with water from the middle of the riffle and transported to the laboratory, where they were refrigerated until the time of processing. The samples were analyzed by the methods outlined in Table 1.

Water samples were collected directly from storm drain outlets II, III, V and VI during two summer rainstorms, of moderate intensity, and during the

TABLE 1.

Water chemistry analysis methods*.

Parameter	Method of Analysis
Alkalinity	Phenolphthalein, bromocresol green titration
Chloride	Argentometric method
Chromium	Colorimetric method
Color	Read at a wavelength of 420 mμ after 10 minutes of centrifugation. A standard curve was drawn using platinum-cobalt color standards.
Conductivity	Y. S. I. Model 31 Conductivity Bridge
Hardness	E. D. T. A. titrimetric method
Nitrogen	
Total	Kjeldahl method
Nitrate	Brucine method
pH	Fisher Accumet Model 520 pH/Ion Meter
Phenol	Chloroform extraction method
Phosphate	
Meta and Poly	Acid digestion
Organic	Total phosphate - (ortho + meta + poly phosphate)
Ortho	Stannous chloride method
Total	Persulphate digestion method
Sodium	Jarrell-Ash Atomic Absorption/Flame Emission Spectrophotometer
Turbidity	Hach Model 2100A Turbidimeter

* Water analysis carried out by Mrs. Gertrude Hutchinson, Biology Technician, Department of Zoology, University of Alberta.

course of routine winter sampling, whenever there was sufficient melting to produce discharge. These samples were handled and analyzed in the same manner as the stream water chemistry samples (Table 1).

Biological

The macrobenthos was sampled with a cylindrical Hess sampler covering an area of 0.085 m^2, and having a net with a 300 micron mesh size. Four samples were collected from each riffle using a stratified random design. Samples were collected from both ends of each riffle and from the shallowest and deepest points within each riffle. When the water level was too low to allow a flow of water through the sampler, a small triangular-mouthed net, having a 300 micron mesh size, was used to collect animals from within the cylinder. The samples were placed in enamel pans, hand picked in the field, and preserved in 95% ethanol. Later, the animals were identified and counted under a dissecting microscope in the laboratory. The dry weight of each taxon was determined by air-drying the specimens in plastic pans at room temperature for 48 hours. The animals were weighed within 3 weeks of collection to minimize any potential weight loss due to storage in ethanol.

A "kick" sample was also taken at each site at the termination of each sampling trip. Kick sampling collected large animals and some rare animals which did not appear in the other samples; these specimens provided a reference source for taxonomic use.

The microbenthos (benthic invertebrates, other than protozoans, capable of passing through a 300 micron mesh net) was sampled with a

core sampler 12 cm^2 in area. Four samples were collected from each riffle using a stratified random design. The cores were brought back to the laboratory unpreserved in glass jars and processed within 6 hours. Samples that were not processed immediately were stored in open jars in the refrigerator. Individuals were identified and counted under low power of a dissecting microscope. Occasionally the amount of silt present made examination of the entire sample impractical; in such cases, subsamples of known volume were removed and counted. The volume of material (clays, silts and fine sands) collected in each core sample was measured to give an estimate of sediment deposition on the stream substrate.

The potamoplankton was sampled with a 160 micron plankton net. Four 3-m hauls were taken from each site. The samples were stored in glass or plastic vials until being brought to the laboratory where they were processed immediately. They were examined under low power of a dissecting microscope, and individuals were identified and counted. The potamoplankton samples were then preserved with 95% ethanol and the ash-free dry weight was determined by drying at 100° C for 24 hours and then ashing at 560° C for one hour in a muffle furnace. The samples were then moistened with distilled water and redried to compensate for waters of hydration lost from clays (Cummins 1962). This provides a measure of the coarse, medium, and some of the fine particulate organic matter (Cummins 1974).

During winter sampling (from December through March), when the stream was ice-covered, the following modifications in procedure were necessary. I sampled areas of 0.5 to 1 m^2 through holes in the ice.

Phenol measurements were disregarded when a chain saw was used to cut the holes because of gasoline spillage. Cylinder sampling was abandoned in favor of semi-quantitative kick samples, during which a known area was extensively worked. These samples were brought to the laboratory in large plastic jugs where they were picked within 8 hours. Samples were stored in the refrigerator with the lids removed until processing. Otherwise, the samples were dealt with in the same manner as the regular cylinder samples. Because of low air temperatures, the cylinder samples of October and November were also brought to the laboratory unpreserved.

Because I could sample only a small area of water in winter, the number of core samples was frequently reduced to two or three. Plankton could not be sampled in winter.

Because the 1976-1977 winter was mild, some of the sites opened earlier than the end of March, and regular "summer" sampling was resumed at such sites as soon as they opened. Site 3 never froze over completely, and winter sampling techniques were only employed here during December. The riffles of sites 4 and 5 opened up sufficiently to allow regular sampling by mid-February. Regular sampling was not resumed at the remaining four sites until April.

RESULTS AND DISCUSSION

Physical-Chemical

Flow

Figure 4 and Appendix 1 present the seasonal averages of physical and chemical constituents in Whitemud Creek. Figure 5 and Appendix 2 show the spatial variations. Winter and summer base flows are both very low, about 0.02 and 0.06 m^3/s, respectively. Flow occurs mainly during periods of high surface runoff, spring snowmelt (from about the last week in March until the end of April), and immediately following heavy rains. Flow also increased during several short periods of thawing in February and early March, 1977. Such thaws and resulting runoff, however, are atypical for the area, and flow usually remains near zero until near the end of March (Erxelben 1972). The response of flow to intense rainfall or sudden thaw can be pronounced. On several occasions, I observed the water level of a riffle to increase from a depth of 2 or 3 cm to more than a meter during a summer thunderstorm. Generally, flow increases at a decreasing rate moving from the edge of the city towards the stream's mouth (Fig. 5). For a given amount of precipitation, more would enter the stream within the city than outside the city, because of greater flow rates across paved surfaces and through the storm sewer systems, and because of the lower potential for groundwater storage in paved surfaces and lawns. This is especially noticeable during heavy rains. Similarly, a greater proportion of the snowmelt can be expected to run into Whitemud Creek within the city (Erxelben 1972), melting being accelerated by the warmer urban

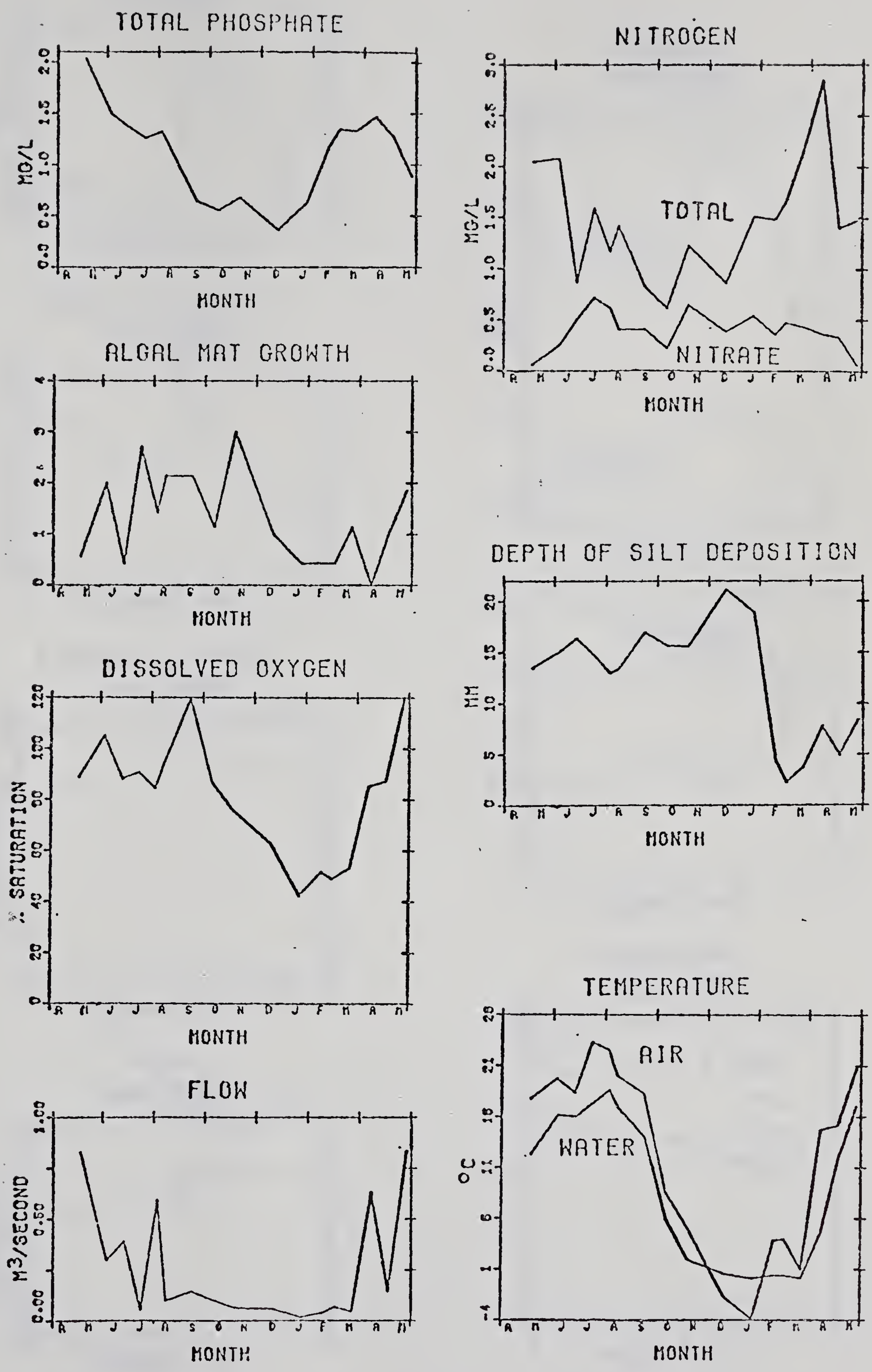


Figure 4. Seasonal variations in chemical and physical constituents, 1976-1977. Algal mat growth is in relative units varying from 0 to 5. An average of all sites is represented for each date.

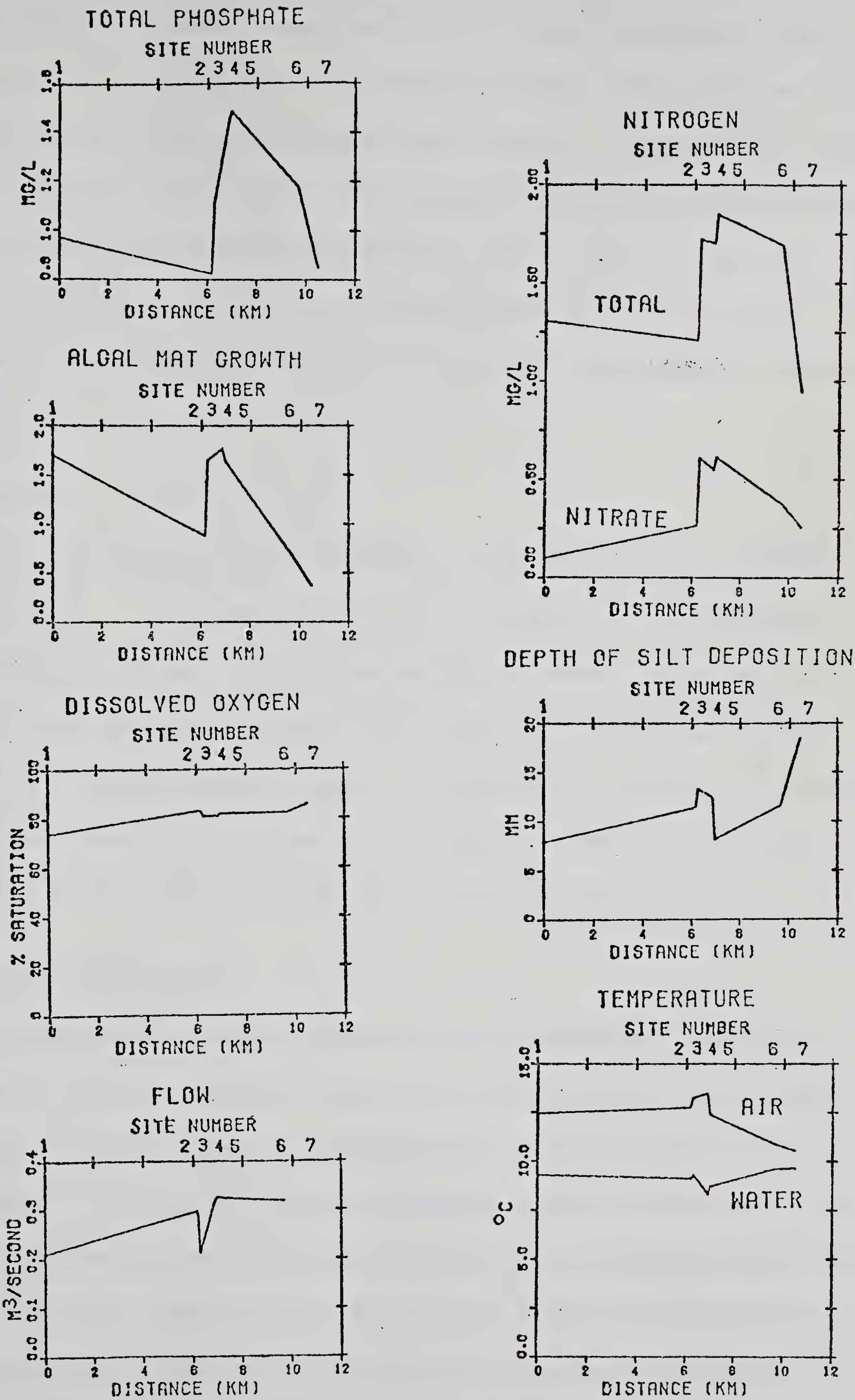


Figure 5. Spatial variations in chemical and physical constituents.
Algal mat growth is in relative units varying from 0 to 5.
Distance is from edge of city moving downstream.
An average of all dates is represented for each site.

environment. However, evapotranspiration will be greater within the city, because of the higher proportion of low albedo surfaces. This would tend to counter increased runoff rates. Also, because of the lower substrate moisture storage capacity within the city, the input from the watershed during dry periods will be less in the city. There are a number of springs along Whitemud Creek within the city, but their input is insufficient to produce the greater flows observed within the city.

Temperature and Precipitation

Air temperatures were maximum in July and minimum in January and March (Fig. 4). There were also some warm spells, causing major thaws, in February. Water temperatures peaked in August, which was one month later than air temperatures; water temperatures remained constant at 0° to 1° C from January to March. There were no apparent differences in water temperature between sites (Fig. 5). Daily precipitation was highest in May, while maximum weekly precipitation occurred in July.

Chemical Constituents

Phosphorus compounds, Kjeldahl (total) nitrogen, particulate organic matter, turbidity, colour, and chromium, all reached peak levels in spring (April or May) and were lowest in winter (Fig. 4, Appendix 1). Their patterns are similar to that of flow and suggest that these substances are being carried into the stream in the runoff, and that their concentrations in Whitemud Creek are dependent upon the runoff volume. This is the same pattern observed by Kluesener and Lee (1974) for phosphate and nitrate concentrations in an urban stream

in Madison, Wisconsin. Meta and organic phosphate also increased in mid-summer during times of heavy rains. One would expect runoff from streets and disturbed areas to produce high concentrations of phosphates, nitrogen, particulate organic matter, colour, and turbidity.

Hardness, alkalinity, conductivity, nitrate nitrogen, sodium, chloride, and phenol concentrations all peaked during the winter and to a lesser extent during the summer (Appendix 1). They reached minimum levels in the spring. They thus reached maximum levels at times of minimum flow. This is the same pattern observed by Wanielista et al. (1977) for conductivity, alkalinity and hardness in a lake receiving urban runoff. It suggests that these substances were present in the runoff in more or less constant concentrations and demonstrated their greatest effect at times of low flow, when dilution was minimum. Chloride in the runoff may arise from road de-icing salts and thus its concentrations would be expected to be highest during a winter such as that of 1976-1977, when there were frequent thaws. Similarly, phenols arise from automobile exhaust emissions and gasoline and oil leakages, which accumulate on the snow. In a normal Edmonton, Alberta winter, without any significant thawing until late March, the chloride and phenol concentrations would probably accumulate in the snow throughout the winter and reach maximum levels in the spring. In milder climates, where winter thaws are typical, winter maxima in chloride concentrations in runoff water have been reported (Oliver, Milne and LaBarre 1974, Field et al. 1975).

The chemical and physical parameters also showed a spatial pattern which generally indicates a relationship with urban runoff.

The basic pattern is a gradual increase in concentrations within the first few kilometers of the city, followed by sharper increases at sites 3 and 5 below the second and third storm drain outlets. Levels decreased between sites 3 and 4 and downstream from site 5. Thus it appears that the major changes in water chemistry along the stream were due to inputs from the first two storm drains, these two storm drains having the largest drainage areas. Downstream from these drains, concentrations of chemical constituents generally decreased, but there were some local increases caused by the inputs from the four remaining storm sewers. This pattern was displayed by nitrogen compounds, hardness, organic phosphate, chloride, conductivity, turbidity, and particulate organic matter (Appendix 2). Total and ortho phosphate, and phenols followed the same pattern except that they did not decrease between sites 3 and 4. This may indicate inputs of these chemicals from the bridge construction site, as well as from the storm sewers.

Sodium and chromium concentrations and pH and alkalinity all decreased more or less consistently as the stream flowed through the city. Chromium is a component of road de-icing salts. Thus its lower concentration within the urban environment might indicate the de-icing salts are not an important constituent of urban runoff entering Whitemud Creek. The decreases in pH and alkalinity may indicate some acidity in the runoff, although the decreases in both are small.

Siltation and Filamentous Algae

Silt deposits were greatest in the winter and least in the spring (Fig. 4). Silt started accumulating on the substrate in early summer and continued to accumulate through winter, and then apparently was washed out by the spring flush. In 1977, the silt deposits were washed out in February, during the atypical early thaws.

Silt deposition generally increases moving downstream within the urban environment (Fig. 5). This pattern probably reflects both the spatial variations in velocities (which determine the rate of deposition) and the spatial variations in silt load.

The percent cover of filamentous algal mats showed a spatial pattern similar to that of phosphate concentration, with maximum growth at sites 3, 4 and 5 (Fig. 5). This could have been because the *Cladophora*, which is very sensitive to inorganic nutrient levels (Hawkes and Davies 1970), is responding to inputs of phosphate from storm sewers. McCracken, Gustafson and Adams (1974) reported the production of algal mats due to nutrient enrichment from storm sewers. Algal mat percent cover did not show obvious seasonal patterns, although there was some indication that percent cover was greater in summer than winter (Fig. 4). The algal mats were frequently washed out by spates produced by heavy summer rains. Within the city the mats were composed primarily of *Cladophora*, with smaller amounts of several kinds of filamentous green and blue-green algae. At site 1, *Spirogyra* was the major component of the mats.

Dissolved Oxygen

Dissolved oxygen values were quite different in the ice-free season than in winter (Fig. 6). During the ice-free season, oxygen was lowest at sites 1, 3 and 4 and highest at site 6. The low dissolved oxygen level at site 1 may be due to decomposition of materials accumulating in a beaver pond at the upstream end of this riffle. Low levels at sites 3 and 4 were probably due to high concentrations of organic materials in the outflow from storm drain 2.

During winter, there was, surprisingly, less oxygen at the control site than within the city. Very low dissolved oxygen concentrations (frequently too low to detect) were also found at a number of other sites upstream from my control site on both Whitemud and Blackmud Creeks by Visscher (1977). I suggest that frequent flow from storm drains in the city during thaws may have aerated the stream water and thus contributed to the high oxygen levels observed in the urban portion of the stream in winter. Also, the riffle at site 3 never froze over completely, and the riffles at sites 4 and 5 both opened up during the first thaw in mid-February. These open areas would result in further aeration of the stream. In contrast, no open water was found upstream from the city during the winter months. The relatively low oxygen levels observed at site 6 probably reflect the effect of the skating rink. The thick ice cover here would prevent aerial mixing, and the rink flooding created a pool-like condition with no perceptable current.

The growth of filamentous algae may be an important source of oxygen in the stream, especially in summer.

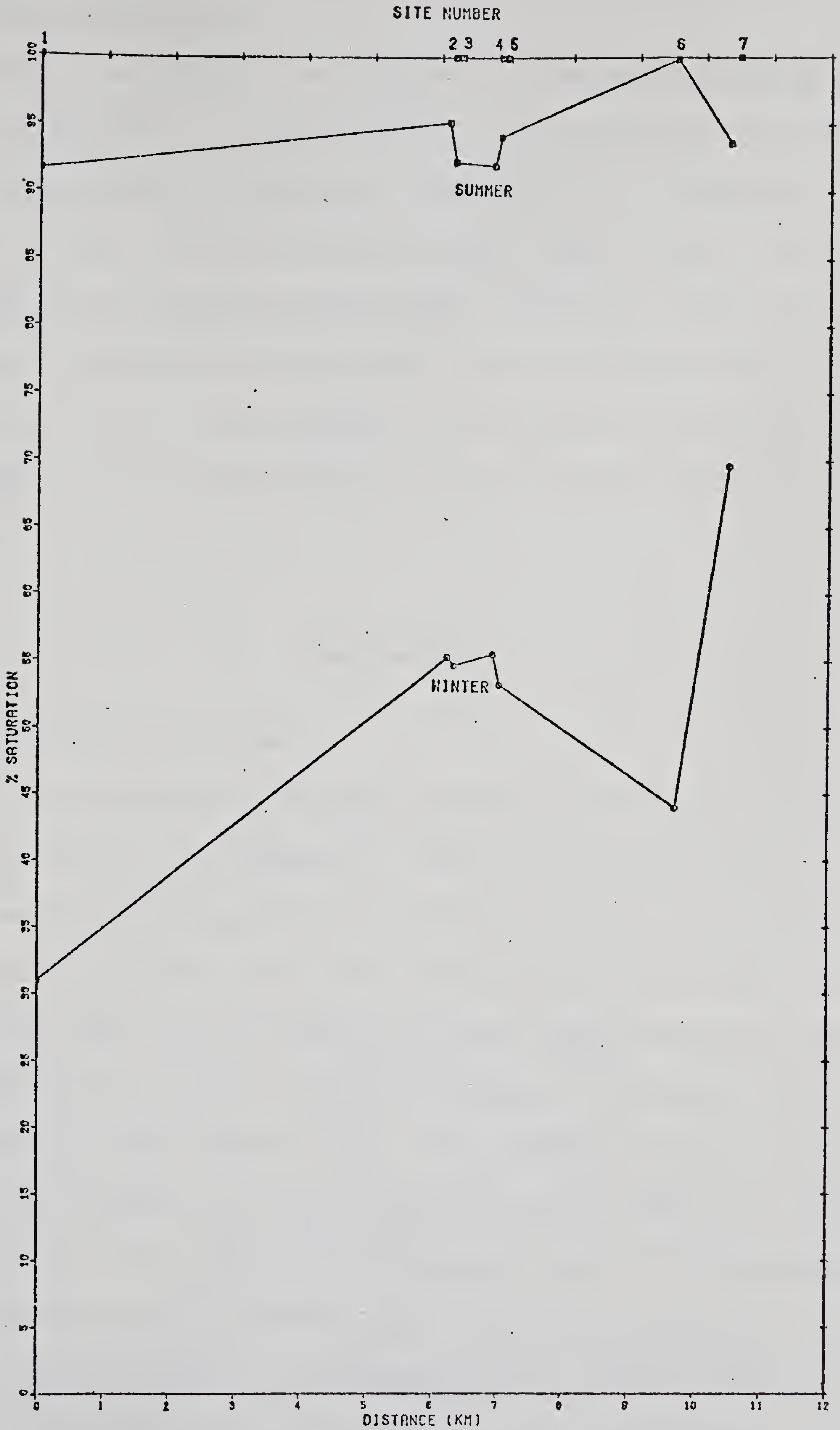


Figure 6. Spatial variations in dissolved oxygen levels in summer and winter. Distance is from edge of city moving downstream. Averages of all summer and winter dates, respectively, are represented for each site.

Storm Sewer Constituents

Physical and chemical constituents of storm sewer discharge are presented in Table 2. Concentrations of total phosphate, Kjeldehl and nitrate nitrogen, conductivity, hardness, colour, turbidity, sodium, chloride, chromium and phenols were higher in storm sewer discharge than in Whitemud Creek itself. Significant inputs of phosphate from urban runoff have also been reported by LaValle (1975) and Mitchell (1975). The measured concentrations of most substances were higher in the runoff from storm sewers II and VI than from sewers III and V.

Macrobenthos

Spacial Distribution of Taxa

The distributions of the macroinvertebrates in Whitemud Creek are shown in Fig. 7 and Appendix 4. Many of the animals (39% of the identified taxa) appear to be adversely affected by the urban environment. Some were never found downstream of site 1 (the control site at the city's edge); others were found further downstream, but in greatly reduced numbers. Some invertebrates that appear to be intolerant of urban conditions are: the amphipods *Gammarus lacustris* and *Hyalella azteca*, the mayflies *Caenis forcipata*, *Baetis* sp. and *Heptagenis hebe*, the caddisflies *Hydropsyche* spp. and *Cheumatopsyche* spp., and the black fly *Simulium* spp. Collectively these taxa represent 65% by number of the macroinvertebrate fauna at site 1. None of these animals, except *C. forcipata*, were ever found

TABLE 2.

Physical and Chemical constituents of storm sewer discharge. All units are in mg/l except color, conductivity (micromhos), pH, phenols (μg/l) and turbidity (JTU's). An average of all samples is presented for each sewer outlet.

	Sewer outlet number				
Constituent	II	III	V	VI	Mean[1]
Alkalinity	4.13	0.94	1.41	1.89*	2.09
Chloride	174*	251	153	133*	178
Chromium	.195*	.165*	.131	.232*	.181
Color	179*	171*	255	198	201
Conductivity	1257*	1034*	638	1861	1198
Hardness					
Total	499*	338	78	771*	422
Calcium	303*	191	48	522*	266
Nitrogen					
Total	4.39*	1.14	1.62	1.81	2.24
Nitrate	0.75*	0.81*	0.47	0.36*	0.60
pH	7.43	7.46	7.33	7.67*	7.47
Phenols	4.46*	5.03	15.3*	8.08*	8.22
Phosphate					
Meta and Poly	0.94*	0.27	0.23*	0.92	0.59
Organic	0.36	0.04	0.28*	0.67	0.34
Ortho	2.83*	0.63	0.96	0.31	1.18
Total	4.13*	0.94	1.41*	1.89*	2.09
Sodium	157*	112	132	211*	153
Turbidity	38.6*	59.4*	30.0	20.8*	37.2

* Indicates levels higher (but not necessarily significantly higher) than is stream water at adjacent sites.

[1] Mean of the 4 sewer outlets.

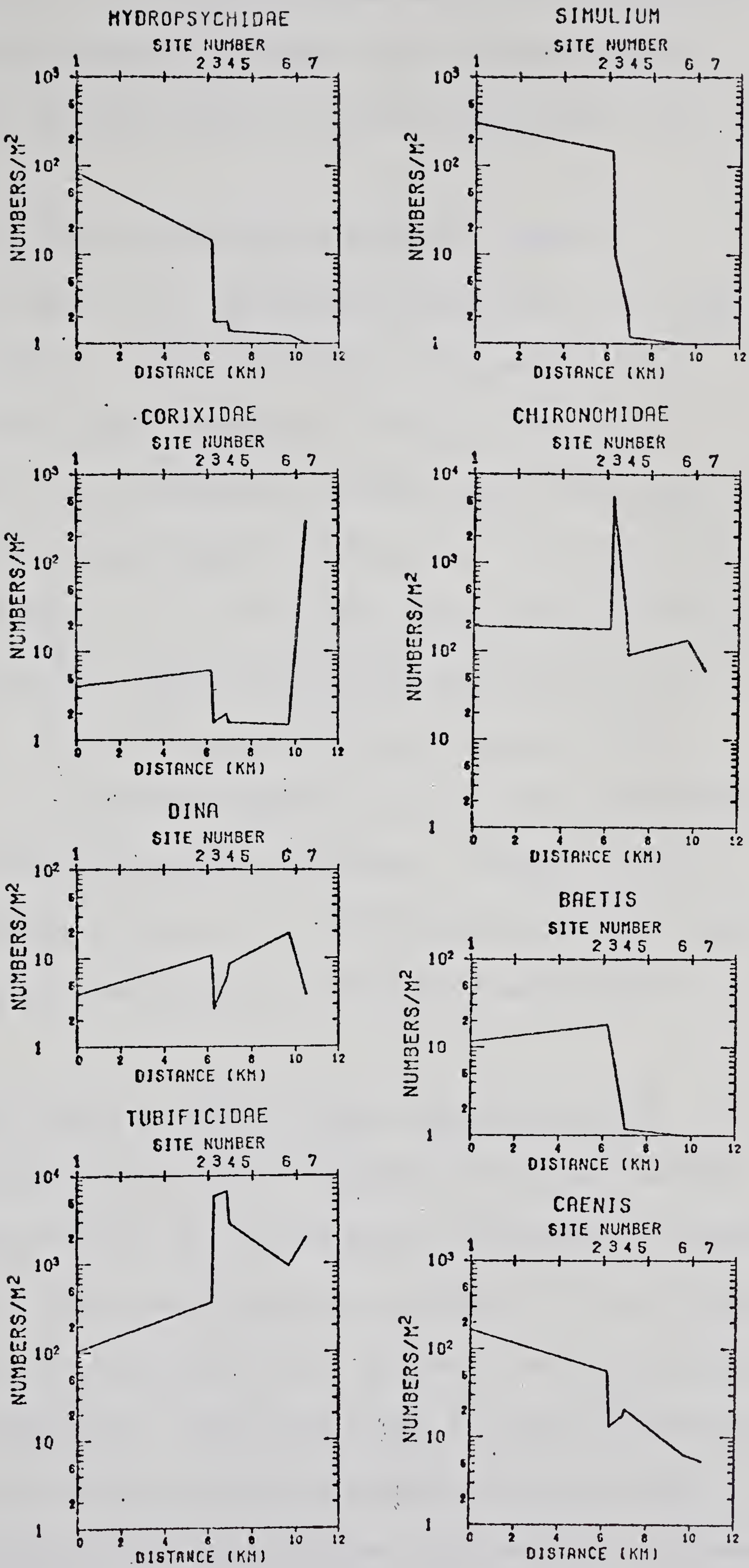


Figure 7. Spatial variations in numbers of some common macroinvertebrates. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

downstream from site 5. It is possible that most of the individuals of these taxa collected below site 2 were ones that had drifted down from upstream, and hence were not normal inhabitants of the urban areas.

The nematodes, leeches and molluscs compose a group of invertebrates that appear to be relatively unaffected by the urban environment (Appendix 4). And a third group, the tubificids and chironomids, exhibited large increases in numbers in the city, especially at sites 3 to 5, below the discharge from storm sewer II. The tubificids and chironomids made up 93% of the total macroinvertebrate fauna at sites 3 through 7. The tubificids alone account for 72% of the fauna within the city. They accounted for only 14% at site 1.

The tubificid fauna of Whitemud Creek is composed of 84% *Tubifex tubifex,* 14% *Limnodrilus hoffmeisteri,* and 2% *L. profundicola* (based on 153 specimens identified to species). *Tufibex tubifex* increased in importance relative to *L. hoffmeisteri* at sites 3 and 5 (below the discharges of storm sewers II and III) and at site 7 (Table 3).

The chironomids found in Whitemud Creek consisted of 18% Tanypodinae, 36% Chironominae (23% Chironomini and 13% Tanytarsini) and 46% Orthocladinae (based on 158 individuals identified to subfamily or tribe). Figure 8 shows the percentage composition of the chironomid fauna at each site. At the control site the fauna was dominated by the Chironomini and Tanypodinae. Within the city, the Orthocladinae became progressively more important and was the dominant group at sites 3, 5 and 6. Thus, it appears that the orthoclads replaced other chironomids

TABLE 3.

Spatial variations in composition of tubificid fauna. The values represent the proportion of each species present at each site.

Species	Site Number 1*	2	3	4	5	6	7
Tubifex tubifex	-	0.57	0.86	0.68	0.86	0.70	1.00
Limnodrilus hoffmeisteri	-	0.29	0.14	0.24	0.14	0.30	0.0
L. profundicola	-	0.14	0.0	0.08	0.0	0.0	0.0
Sample size	0	7	36	25	8	23	54

* No tubificids from site 1 were identified to species because of the low numbers of individuals, especially mature ones, collected there.

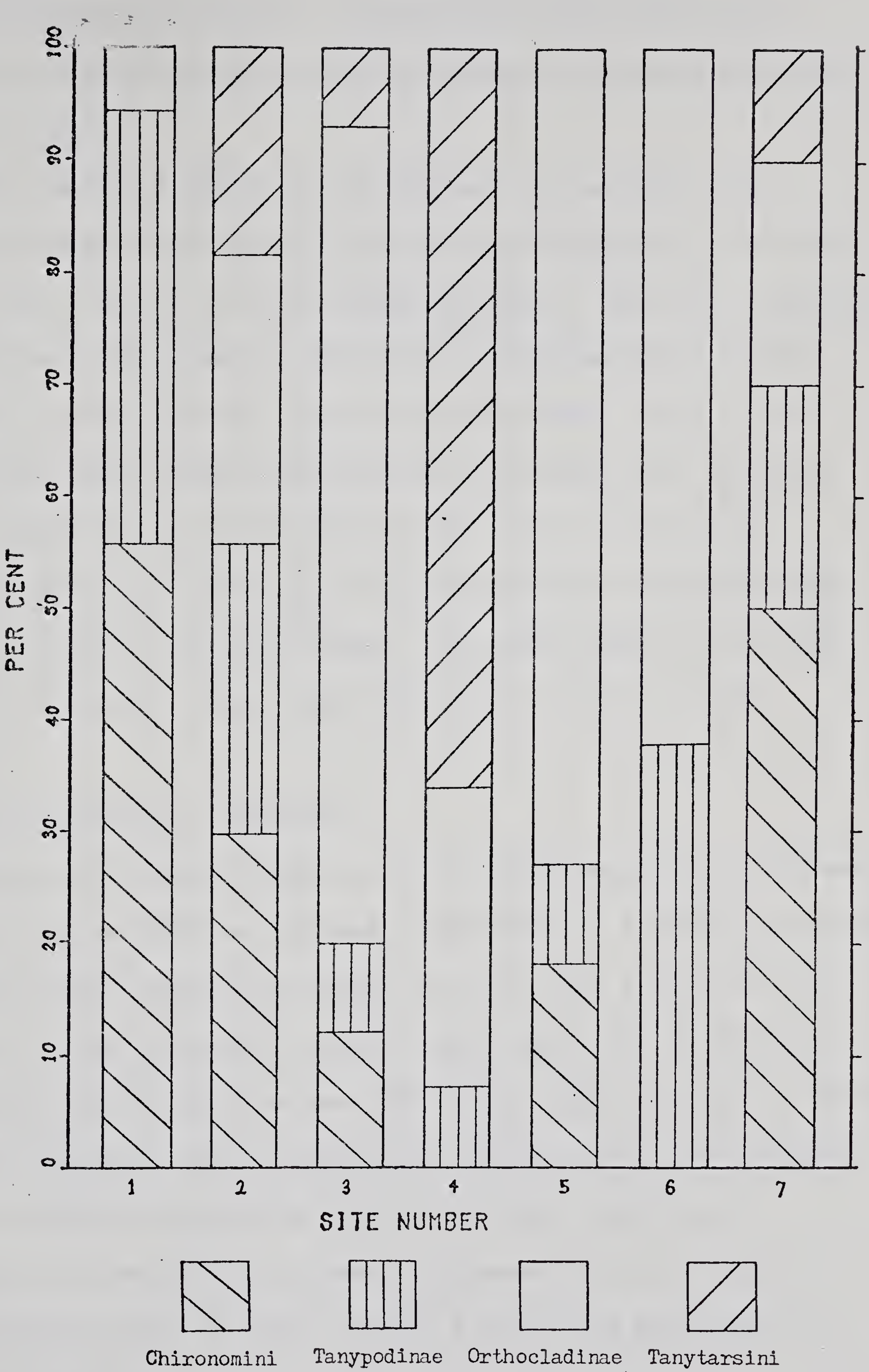


Figure 8. Spatial variations in chironomid faunal composition. An average of all dates is presented for each site.

within the urban environment. The Tanytarsini were also able to survive at some of the urban sites and actually dominated chironomid fauna at site 4.

The stoneflies *Zapada* sp., *Brachyptera* sp. and *Capnia* sp., the mayfly *Ephemerella inermis*, and the snail *Helisoma* sp. were all found only at site 7 and only during the winter. The three stoneflies and the mayfly are common in the North Saskatchewan River a short distance upstream from the mouth of Whitemud Creek. Thus it is likely that these animals moved into Whitemud Creek from the river. This could easily be facilitated in winter when the waters of Whitemud Creek were backed up by the river and high dissolved oxygen levels occurred in the urban stream. The large numbers of corixids found at the mouth at this time probably had a similar origin.

Spatial Distribution: Diversity

Macroinvertebrate diversity was calculated according to Simpson's formula (Simpson 1949) as outlined in Appendix 5. Diversity generally decreased moving downstream through the city, with a sharp drop at site 3, and some recovery at sites 4 and 5 (Fig. 9). The decrease in diversity in the city was due both to the large increases in tubificid numbers, leading to their dominating the fauna, and to the disappearance or decline in numbers of many of the other taxa (Fig. 7). Richness, or number of taxa present, decreased from an average of nearly 16 per sample at site 1 to only 6 to 10 taxa per sample in the city (Fig. 10).

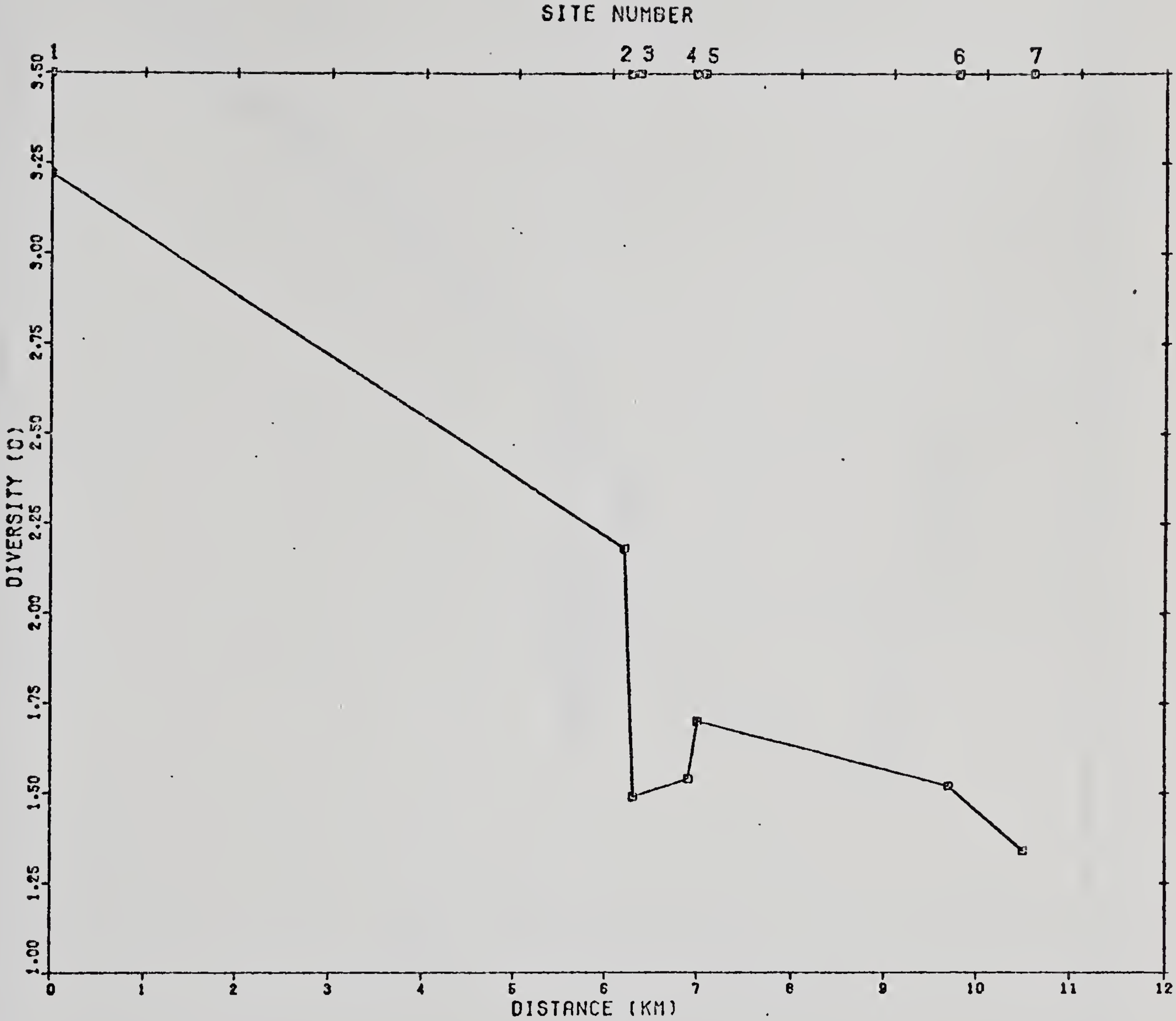


Figure 9. Spatial variations in macrobenthic diversity. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

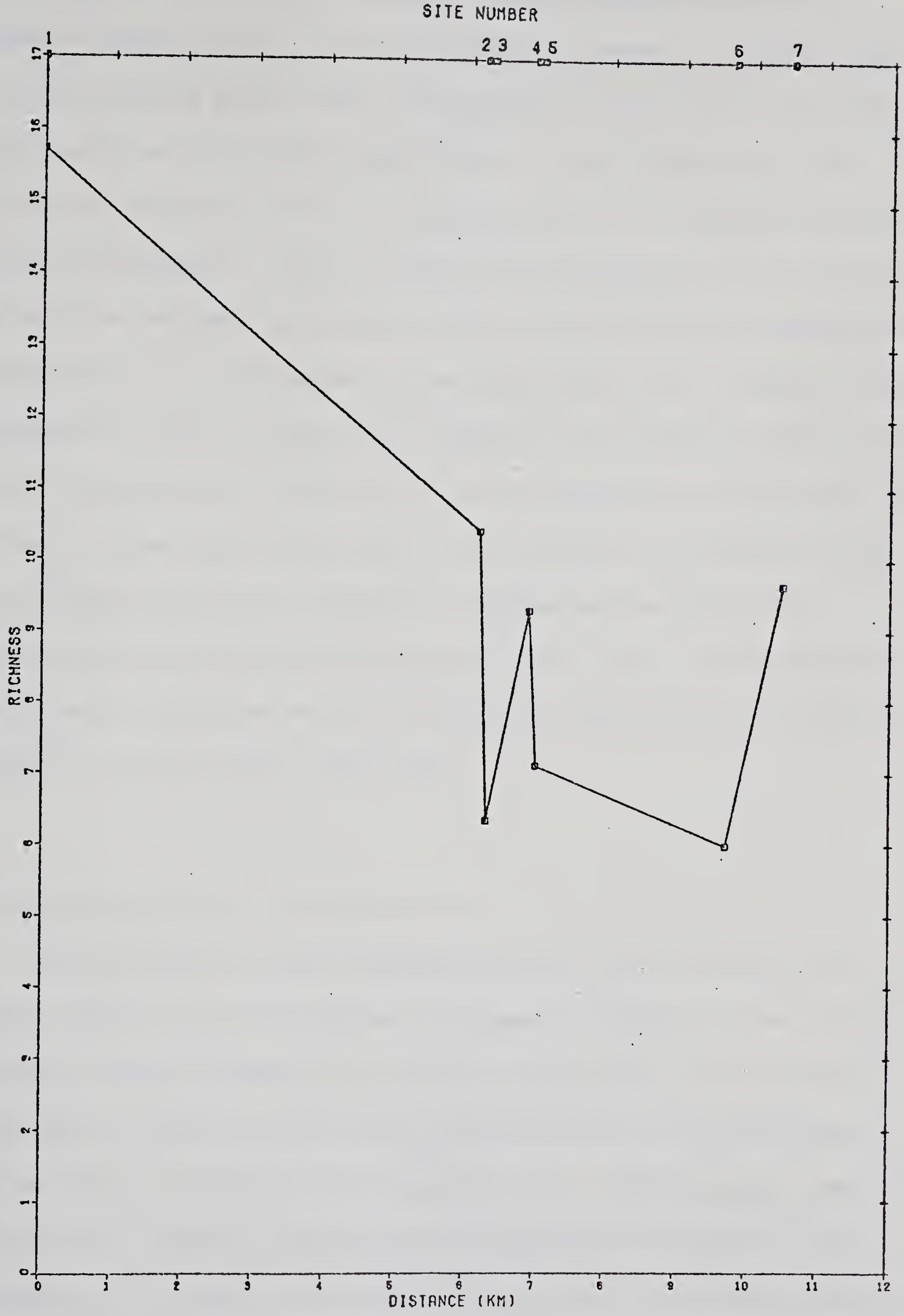


Figure 10. Spatial variations in macrobenthic richness. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

Storm sewer discharge appears largely responsible for the observed faunal changes. The very sharp decreases in diversity and richness between sites 2 and 3 corresponded to the point of discharge from storm sewer II. This sewer drains a very large area. The increases at sites 4 and 5 were apparently due to recovery from this severe disturbance. Thus the effects of storm drain II were severe enough that recovery proceeded in spite of the Whitemud Freeway Bridge construction and the discharge from storm sewer III. Although diversity increased at site 5, there was a reduction in richness at that station, indicating that some of the rarer taxa may have been eliminated by the effects of the third storm sewer. The increase in richness at site 7 was probably due to the occasional upstream movement of a few individuals of some North Saskatchewan River taxa. Low diversities are usually associated with fluctuating or unfavourable environmental conditions (Pielou 1967, Zand 1976).

Spatial Distribution: Standing Crop

Total density of the macroinvertebrate fauna is shown in Fig. 11. Total density was most affected by changes in numbers of the two most abundant taxa, the Tubificidae and the Chironomidae. The ten-fold increase in density at site 3 was apparently due to the enriching nature of the discharge from storm sewer II. The subsequent downstream decreases in density represented recovery from the effects of this discharge. The high total density at site 7 was due almost entirely to corixids that moved into this station from the North Saskatchewan River in winter.

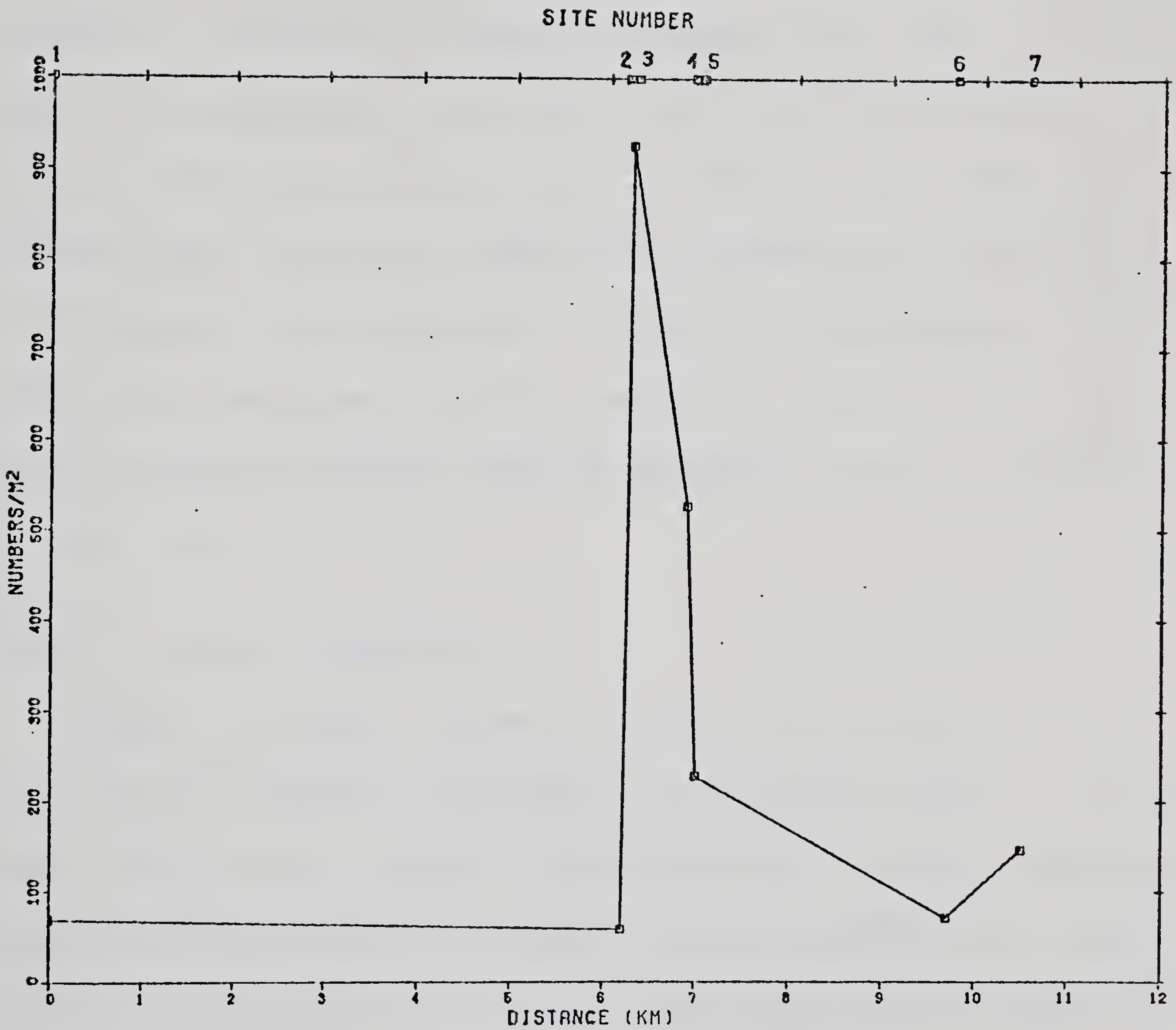


Figure 11. Spatial variations in macrobenthic density. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

The maximum recorded density of tubificids in Whitemud Creek was $66,000/m^2$, in May 1977. This is about one third the tubificid density recorded by Gaufin and Tarzwell (1956) for a stream grossly polluted by organic sewage. Brinkhurst and Kennedy (1965) found more than one million tubificids per square meter in a grossly polluted stream in Great Britain.

The spatial variation in total macroinvertebrate biomass is shown in Fig. 12. Tubificids accounted for almost half (48%) of the standing crop. Chironomids (14%), *Dina* spp. (8.3%) and *Caenis forcipata* (3.1%) also made important contributions. The other 35 taxa made up the remaining 26%, with none of these taxa accounting for more than 3% of the biomass each (Appendix 6). There was less biomass at all but two of the urban sites (4 and 7) than at the control site. Thus the macroinvertebrate standing crop was apparently reduced by the influence of urban runoff.

Causes of Spatial Distribution

Organic enrichment appears to be the most important factor influencing the spatial distribution of macroinvertebrates in Whitemud Creek. The two very abundant tubificid species, *Tubifex tubifex* and *Limnodrilus hoffmeisteri*, are those usually associated with organic enrichment (Brinkhurst 1965, Whitely 1968, Aston 1973). Many of the animals that were reduced in numbers in the city are reported as being very intolerant of much organic pollution, e.g. *Gammarus* (Surber 1953, Butcher 1955, Hawkes and Davies 1970, Nuttall and Purves 1974), *Heptagenia* (Butcher 1955), *Hydropsyche* (Butcher 1955, Learner et al. 1971), *Hexatoma* (Paine and Gaufin 1956) and the simuliids (Surber 1953). And those invertebrates found generally distributed within the stream

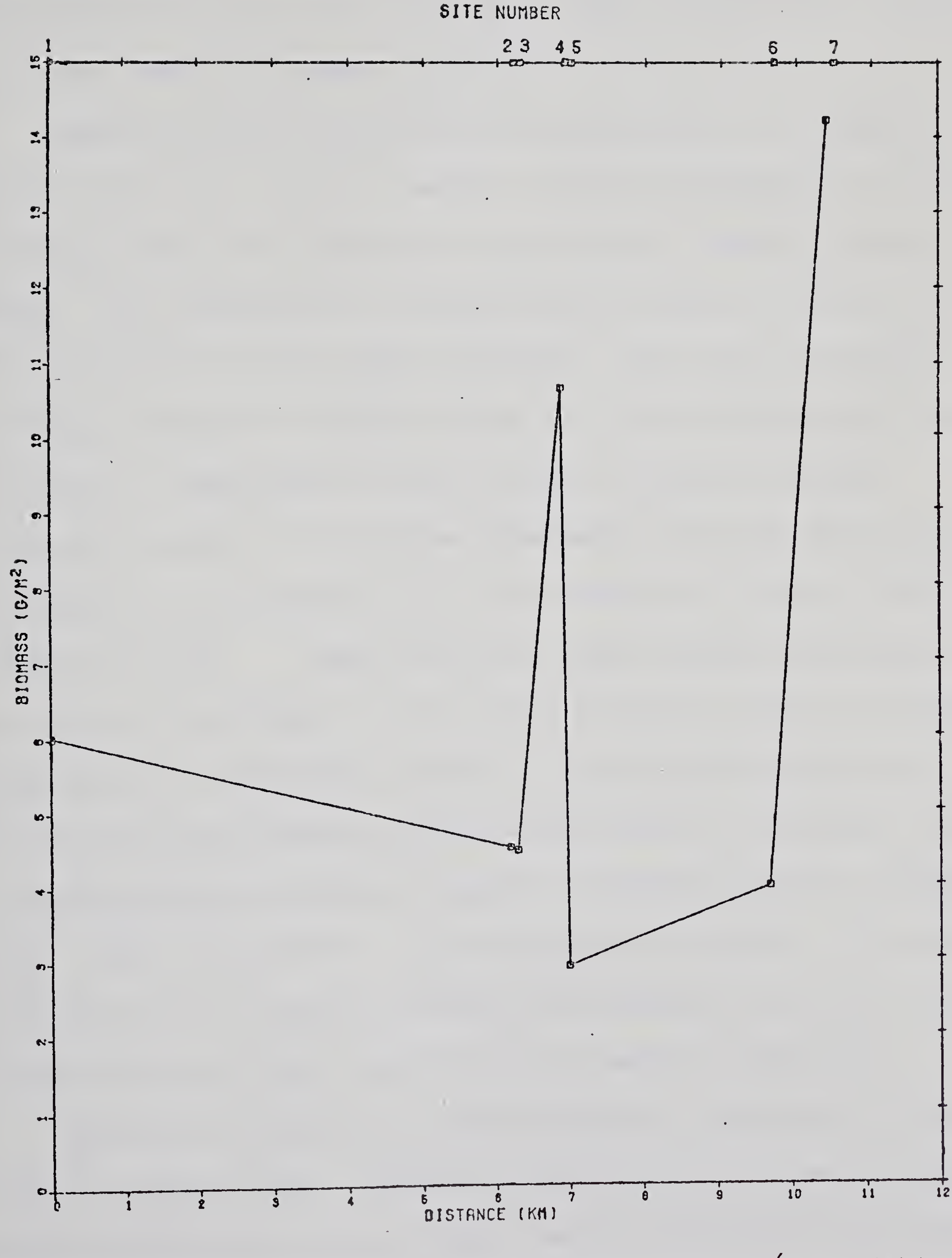


Figure 12. Spatial variations in macrobenthic biomass (dry weight, averages of all sampling dates). Distance is from edge of city moving downstream. An average of all dates is represented for each site.

are apparently tolerant, e.g. leeches, molluscs (*Physa, Lymnaea, Ferrissia* and *Pisidium*), chironomids, and air-breathing corixids and dytiscids (Butcher 1955, Surber 1953, Hynes 1960). These kinds of faunal changes, associated with enrichment, are consistent with reports from different parts of the world (Hynes 1960) and have been reported in a large number of studies (Table 4).

Organic material entering the stream would tend to be deposited on the stream bed, with the resultant bacterial decomposition producing an oxygen demand and ultimately an oxygen deficit (Hawkes and Davies 1970). Thus animals would be removed from the fauna on the basis of their intolerance of low oxygen conditions. Under the most extreme conditions observed in Whitemud Creek, only the tubificids were able to survive. Tubificids are able to thrive in grossly polluted situations because of the abundant food source available and the lack of competitors and predators,plus their respiratory features (Whitely 1968, Aston 1973). Oxygen conditions improve moving downstream from the source of enrichment, as the organic material is used up or diluted. This allows the chironomids, which are competitors (Brinkhurst and Kennedy 1965) and possible predators (Loden 1974) of tubificids, the erpobdellid leeches, which are important tubificid predators (Brinkhurst 1964, 1965; Aston 1973), and the molluscs to survive. Thus the diversity of the community increases, both because of the more varied fauna and because of the reduction in the number of tubificids.

Thus the low diversities observed should be associated with low dissolved oxygen levels. However, in Whitemud Creek, dissolved oxygen concentrations were never much lower, and were sometimes even higher, at the sites where the lowest diversities were observed. I suggest that

TABLE 4.

Literature reporting effects of enrichment on stream invertebrates.

Aston, 1973.

Brinkhurst, 1965.

Brinkhurst, 1966.

Brinkhurst and Kennedy, 1965.

Burlington, 1962.

Egloff and Brakel, 1973.

Gaufin, 1958.

Gaufin and Tarzwell, 1956.

Goodnight, 1973.

Hawkes and Davies, 1970.

Learner et al., 1971.

Nuttall and Purves, 1974.

Olive and Dambach, 1973.

Olive and Smith, 1975.

Surber, 1953.

this is probably due to the large mats of *Cladophora* present. The algae photosynthesize during the day (which is when the water samples for oxygen determination were collected), thus contributing oxygen to the system, possibly in large enough amounts to counteract diurnal bacterial decomposition. But at night, algal respiration would contribute to oxygen demand in the stream. Large dissolved oxygen fluctuations, associated with blooms of *Cladophora*, have been reported by Gaufin (1958), Gaufin and Tarzwell (1956), Hawkes and Davies (1970) and Davis (1975).

Siltation is probably also involved in the observed faunal changes. Silt deposition showed a fairly constant increase moving downstream within the city while diversity decreased. Animals such as *Caenis*, *Pisidium*, snails and the tubificids are characteristically tolerant of silt deposition (Longford and Bray 1969, Nuttall 1972, Nuttall and Purves 1974, Barton 1977). Leeches and the limpet *Ferrissia rivularis*, both found throughout Whitemud Creek, are also reported to be tolerant of siltation, although the latter is fairly intolerant of organic pollution. The disappearance of filter-feeding invertebrates also suggests that siltation is affecting the spatial distribution of Whitemud Creek invertebrates. For example, both *Simulium* and the hydropsychids are intolerant of siltation (Hamilton 1961, Chutter 1969, Nuttall and Purves 1974, Barton 1977). The Tanytarsini, which are common at some urban sites, have previously been associated with siltation (Aggus 1971). Their dominance in the chironomid fauna at site 4 suggests that there may be increased inputs of silt from the Whitemud Freeway Bridge construction at this site. The dominance of the urban chironomid fauna

by Orthocladinae, rather than Chironominae which are usually associated with low oxygen conditions, may also indicate siltation in addition to the nutrient enrichment. Learner et al. (1971) found that orthoclads dominated the fauna in the River Cynon, South Wales, where there was both high siltation and organic enrichment.

Hamilton (1961) showed that it is the deposition of silt, rather that the silt load or turbidity, that affects the benthic fauna. Silt acts to reduce the heterogeneity of the physical environment and eliminate microhabitats by filling in the interstices between gravel particles within the riffle as well as interfering with filter-feeding and respiration. I do not know whether the silt being deposited in Whitemud Creek is from storm drain effluent, the bridge construction site, or increased bank erosion caused by the more intense spates, although all three might be involved.

Siltation alone cannot account for the observed changes. Dominance of the tubificid fauna by *Tubifex tubifex* and *Limnodrilus hoffmeisteri* and the growth of *Cladophora* mats are almost certainly evidence of enrichment. And in areas of high siltation alone, the density of the total fauna is usually reduced. Also, animals normally associated with *Cladophora* in areas of high siltation, such as naidids, *Baetis* and *Ephermerella* (Hynes 1960), were absent or rare in the most altered areas of Whitemud Creek. Thus it appears that both enrichment and siltation are affecting the macroinvertebrate fauna of Whitemud Creek. Inputs of organic matter, inorganic nutrients, and silt frequently arise from a common pollution source (Brinkhurst 1965, Hawkes and Davies 1970). Street dust, carried by the storm sewers, can be expected to contribute to all three.

The faunal changes do not fit the patterns likely to result from the discharge of toxic substances. Toxic material generally eliminates some stream taxa altogether and reduces the numbers of those surviving (Surber 1953). Pollution by heavy metals, such as lead, is probably not significant in Whitemud Creek. The dominant animals of the urban reaches, the tubificids, leeches, and molluscs are among those that have been reported to be the most sensitive to heavy metals (Aston 1973, Brković-Popović and Popović 1977). And *Cladophora* is also very sensitive to heavy metals (Hynes 1960).

Increased salinity or inputs of pesticides could eliminate the kinds of animals that disappear from urban Whitemud Creek. However, tubificid and chironomid numbers increased in the city. And, while these animals are more tolerant of pesticides than many others, they are nevertheless adversely affected (Whitten and Goodnight 1966). Also, the presence of pesticides should be associated with a diverse oligochaete fauna, rather than dominance by *Tubifex tubifex* and *Limnodrilus hoffmeisteri* (Brinkhurst 1966). And G. Griffiths (pers. comm.) has demonstrated that adulticiding for mosquitoes in the Whitemud Creek valley does not affect the fauna within the stream. The presence of coleopterans and *Caenis* in Whitemud's urban reaches is also inconsistent with pesticide pollution, and there is no chemical evidence of increased salinity. Conductivity rises only slightly within the city and sodium concentrations actually decrease.

The kinds of faunal changes encountered are also inconsistent with those expected from chlorine, ammonia or cyanide poisoning. Thus, it appears that the toxic substances usually reported from urban runoff are not present in high enough concentrations in Whitemud Creek to have an important effect on the macroinvertebrates.

Temporal Distribution

There was a great deal of seasonal fluctuation in macroinvertebrate diversity (Fig. 13). These fluctuations may be the results of particular runoff incidents. However, because of the difficulties in assessing the importance of the various incidents, it is not possible to show any correlations. Nevertheless, diversity appeared to be generally higher in winter than in the ice-free season. The apparent increase in diversity in winter was likely due to a reduction in tubificid numbers. Most of the recruitment to the tubificid populations occurs during the ice-free season (Brinkhurst and Kennedy 1965), and thus numbers would decline in winter, causing an increase in measured diversity. Other studies have also reported winter maxima in diversity that can be associated with the life cycles of the animals involved (Mackay and Kalff 1969, Hawkes and Davies 1970, Mackey 1977). Slobodchikoff and Parrott (1977) reported lower diversities in winter, but this was due to life cycle features of one caddisfly species, which dominated the fanua of this Arizona stream.

While diversity increased in the winter, richness decreased at all sites (Table 5). This further suggests that winter diversity increase was due almost entirely to a decrease in tubificid numbers and not to an overall increase in the quality of the fauna. There was a more or less continuous decline in diversity from March through May, when snowmelt runoff was entering the stream. A spring reduction in diversity was also reported from an organically enriched stream in Oklahoma by Wilhm and Dorris (1968).

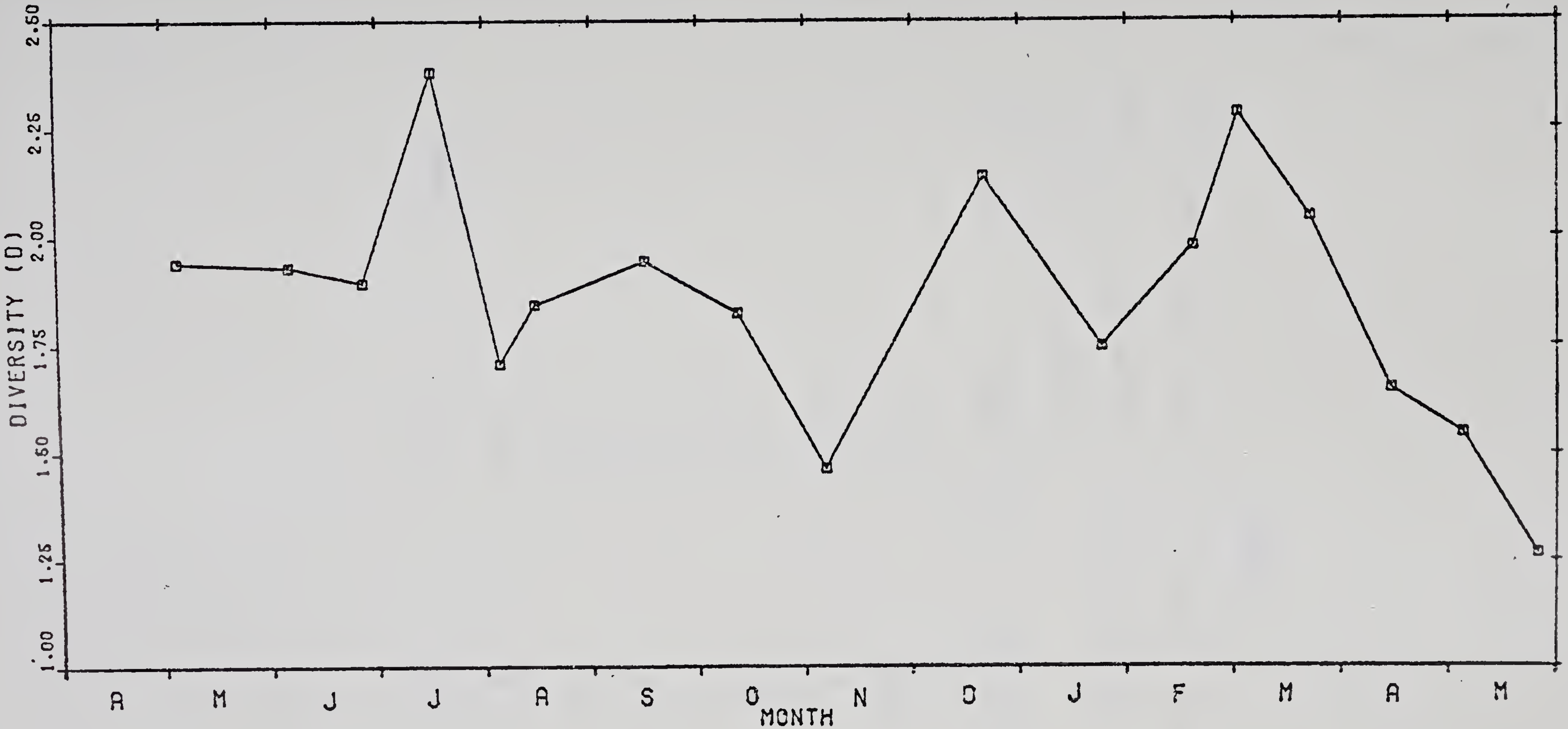


Figure 13. Seasonal variations in macrobenthic diversity, 1976-1977. An average of all sites is represented for each date.

TABLE 5.

Macroinvertebrate taxonomic richness in summer and winter. Averages of all summer and winter dates, respectively, are presented for each site.

Site number	Summer mean	Winter mean	Decrease
1	18.1	10.0	45%
2	12.1	4.4	66%
3	7.4	3.8	49%
4	10.8	5.6	48%
5	8.0	5.0	38%
6	7.2	3.2	56%
7	10.8	6.8	37%
Mean	10.6	5.5	48%

High total macroinvertebrate densities were associated with low diversities (Fig. 14). There were increases in numbers in the spring, particularly in 1977, relatively low densities in late winter and to a lesser extent in late summer. Density changes, at least in part, appear to reflect the life cycles of the dominant animals. For example, the tubificids reproduced throughout the ice-free season, but peak tubificid reproduction apparently occurred in spring.

Indicator Organisms

The most abundant invertebrate taxon was the Tubificidae, making up 72% of the numbers and 49% of the biomass of all the animals collected. The percentage (of the total fauna) numbers and biomass of tubificids provided a good indication of the overall condition of the fauna, and in particular of the diversity, at any site and time. King and Ball (1964) have also found that the weight of tubificids relative to that of the rest of the fauna is a good indicator of stream pollution.

Brinkhurst (1966, 1967) and Learner et al. (1971) have shown that the ratio of the numbers of *Limnodrilus hoffmeisteri* to the numbers of *Tubifex tubifex* increases with increasing organic pollution. However in Whitemud Creek, I found the proportion of *T. tubifex* to *L. hoffmeisteri* to increase in response to urban runoff inputs (Table 3).

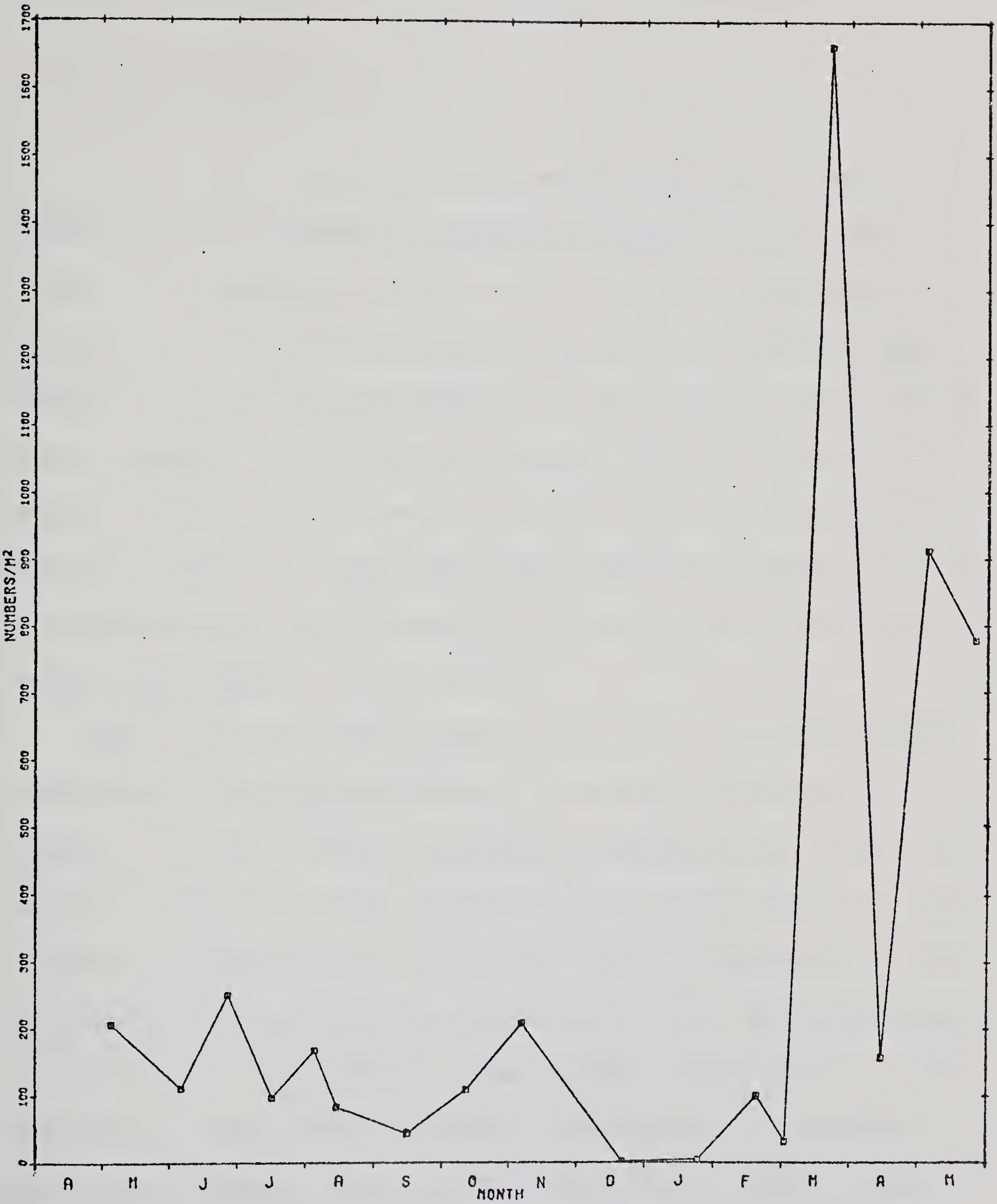


Figure 14. Seasonal variations in macrobenthic density, 1976-1977. An average of all sites is represented for each date.

Amphipods (*Hyalella azteca* and *Gammarus lacustris*), hydropsychids (*Hydropsyche* spp. and *Cheumatopsyche* spp.) and *Caenis forcipata* were good indicators of unaffected conditions in Whitemud Creek.

Life Cycle of *Caenis forcipata*

I studied the life cycle of one macroinvertebrate in detail. *Caenis forcipata* McDunnough was the most abundant mayfly in the stream. The following measurements were made on all individuals of this species collected in the cylinder and kick-net samples: head width (the maximum distance between the outer margins of the compound eyes), abdomen width (the maximum distance across the abdomen measured from the tips of the postero-lateral spines, ignoring gill opercula), and total length (body length excluding antennae and cerci). Although no adults were collected in the field, nymphs were reared to the adult stages in the laboratory.

Eggs were laid in late July and early August and hatched almost immediately. Nymphs grew throughout autumn until freeze-up in November (Fig. 15). Little or no growth occurred during winter. In Whitemud Creek and in other Alberta streams (H. Hamilton, pers. comm.) *Caenis*'s guts were empty during winter. Growth resumed early in April when the stream became ice-free and continued until the adults emerged in late July and early August. Since no adults were observed in the field (and I could not use emergence traps becuase of vandalism), emergence is inferred from the time when the large nymphs disappeared. The appearance of a number of small nymphs in April suggests that delayed hatching might be extensive for this species.

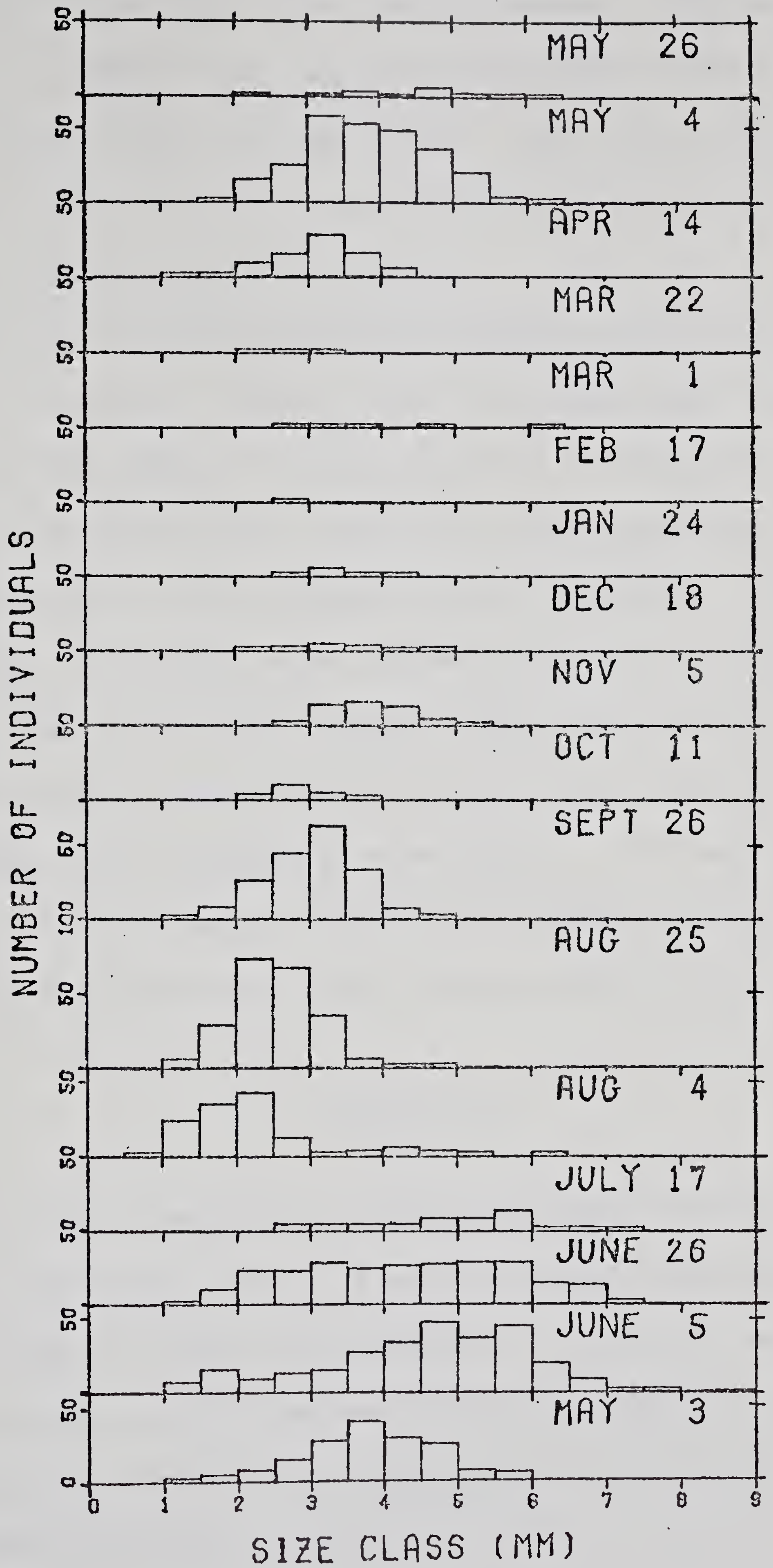


Figure 15. Length-frequency histograms of *Caenis forcipata*, 1976-1977. All animals collected from all stations are included.

I could collect only a few nymphs in winter. This was due mainly to the nymphs being collected from a smaller area in winter, because I had to sample through the ice. Also some of the nymphs might have moved deeper into the substrate or migrated to the pools, and hence avoided collection in winter.

There were no obvious differences between sites in the timing of hatching or emergence. However some sites consistently support larger individuals than others. The mean length of *C. forcipata* nymphs was greater at sites 2 and 3 than at the other sites (Fig. 16). This suggests that there is greater growth of nymphs at some sites in the city. The average annual apparent growth rate (change in mean size over time at each site) showed a similar pattern (Fig. 17). The greater growth rates observed at sites 3 to 5 are apparently due to enrichment from discharge of storm drain II. The negative growth rates are due to emergence of adults and hatching of young nymphs, because these two phenomena reduce the mean size of nymphs.

Microbenthos

The spatial distribution of the microbenthic (benthic invertebrates other than protozoans capable of passing through a 300 micron mesh net) fauna is shown in Figure 18 and Appendix 7. Tubificids were the dominant animals, making up 30% of the microbenthos by numbers. The small tubificids were found throughout the stream, increasing in density moving downstream within the city.

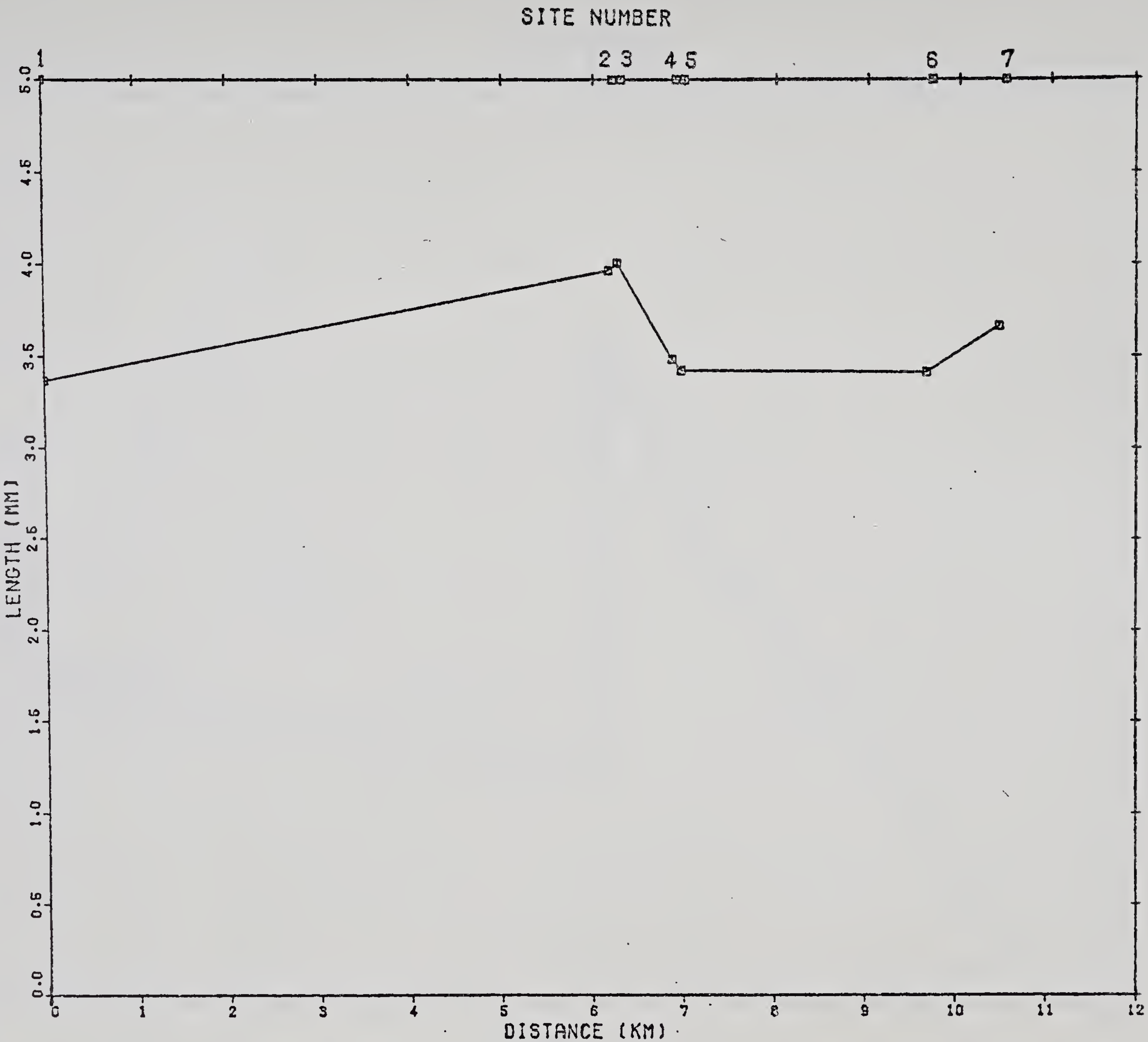


Figure 16. Spatial variations in the mean size of *Caenis forcipata* nymphs, 1976-1977. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

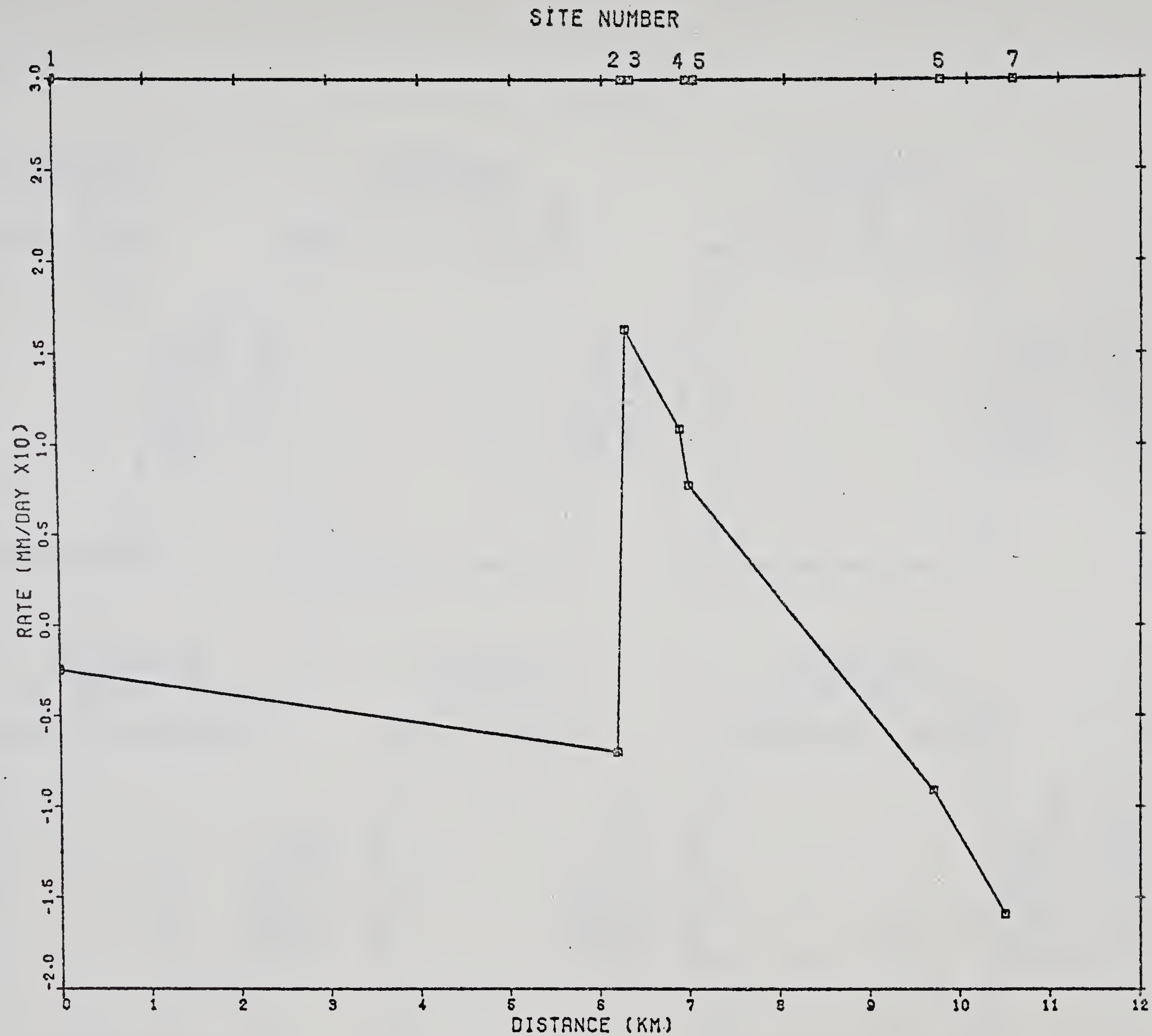


Figure 17. Spatial variations in the growth rate of *Caenis forcipata* nymphs. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

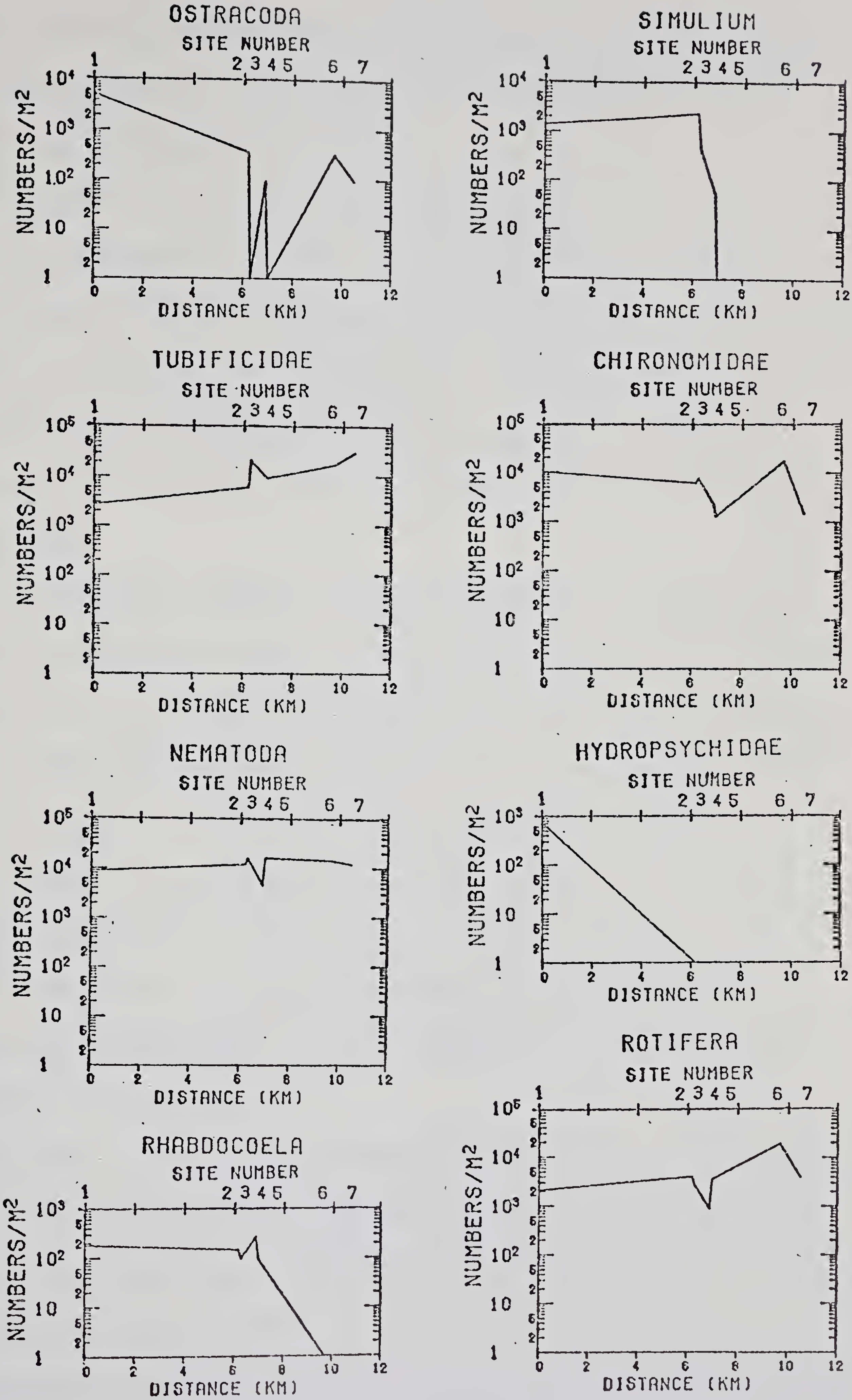


Figure 18. Spatial variations in numbers of some common microinvertebrates. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

Early instar chironomid larvae, nematodes and rotifers (mostly creeping bdelloids) were also common in the microbenthos (Fig. 18). Nematodes were generally distributed.

Of the other 11 recorded taxa of microbenthos, 5 were found primarily or exclusively at sites 1 and 2 (Appendix 7). Two taxa were found mainly near the mouth, and the remaining 4 taxa were generally distributed or showed no obvious patterns. Ostracods, early instar hydropsychids, *Caenis forcipata*, *Baetis* sp. and *Simulium* spp. are all good microbenthic indicators of relatively undisturbed areas of Whitemud Creek.

The microbenthic richness had a spatial pattern similar to that of the macrobenthos (Fig. 19). Once again decreases in richness corresponded to the points of discharge of the first two storm sewers, and hence urban runoff appears to be the cause of the observed faunal changes. Microbenthic diversity followed the same pattern as richness, except for the decrease in diversity at site 7 (Fig. 20). As for the macrobenthic diversity, the increased richness at site 7 was likely the result of upstream movement of a few animals from the North Saskatchewan River. This increases the richness, but not the diversity of the fauna.

The micro- and macrobenthic patterns within the city were similar (Fig. 21). The main difference was the recovery in the microbenthos at site 6. This may indicate that the microbenthic fauna is slightly less sensitive to urban runoff.

Microbenthic densities were greater than those in the macrobenthos. Table 6 compares tubificid micro- and macrobenthic densities. The

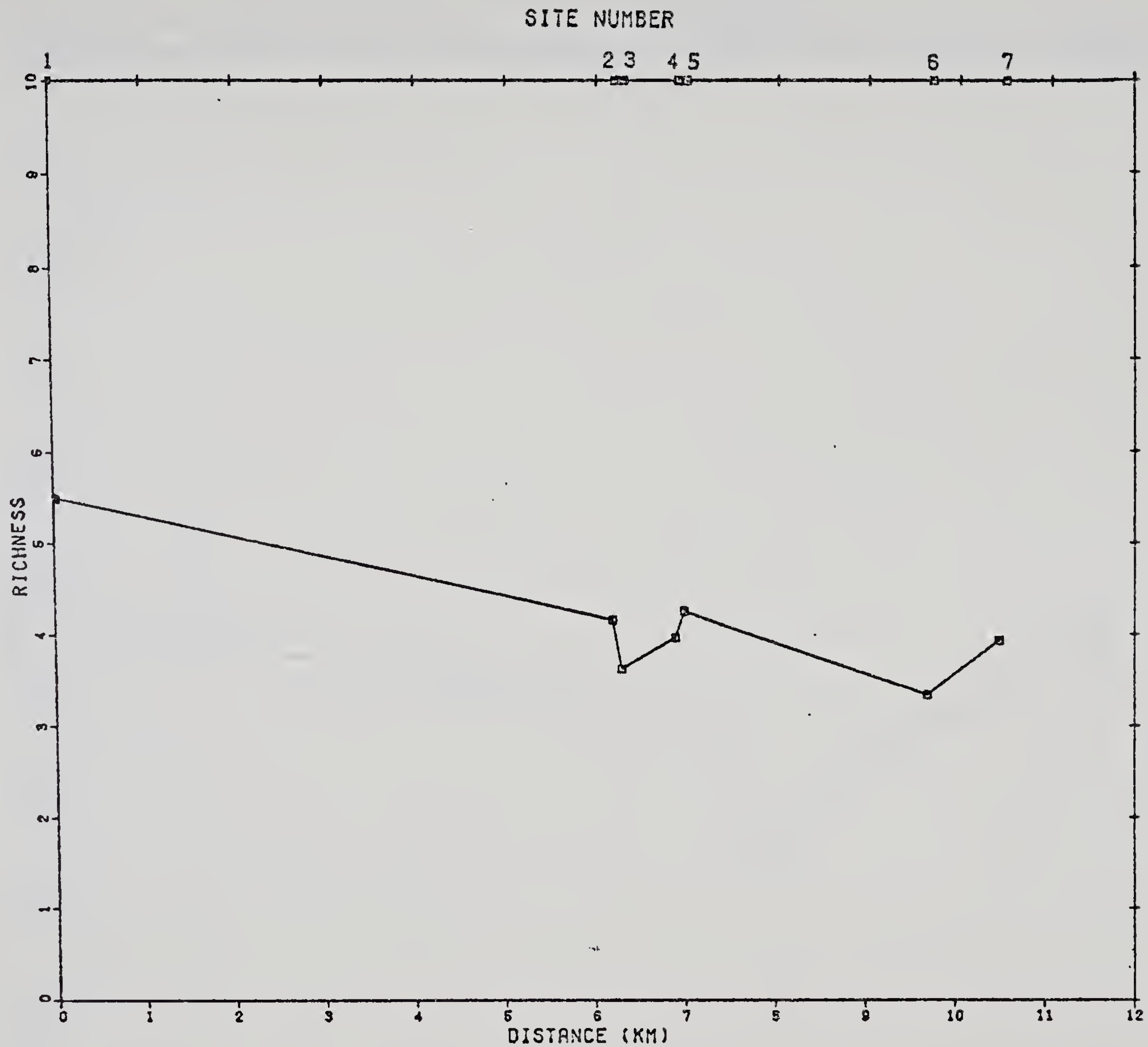


Figure 19. Spatial variations in microbenthic richness. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

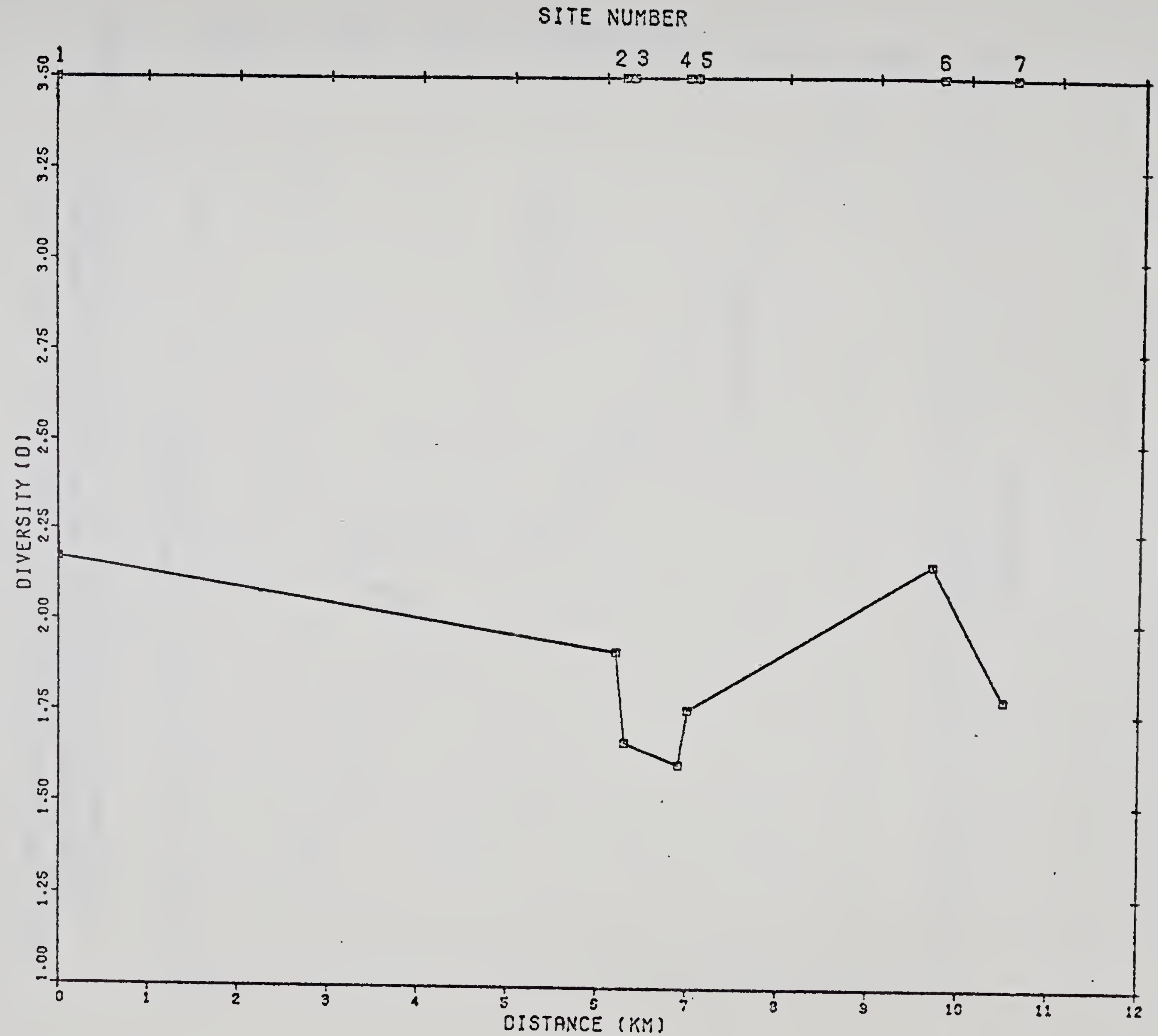


Figure 20. Spatial variations in microbenthic diversity. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

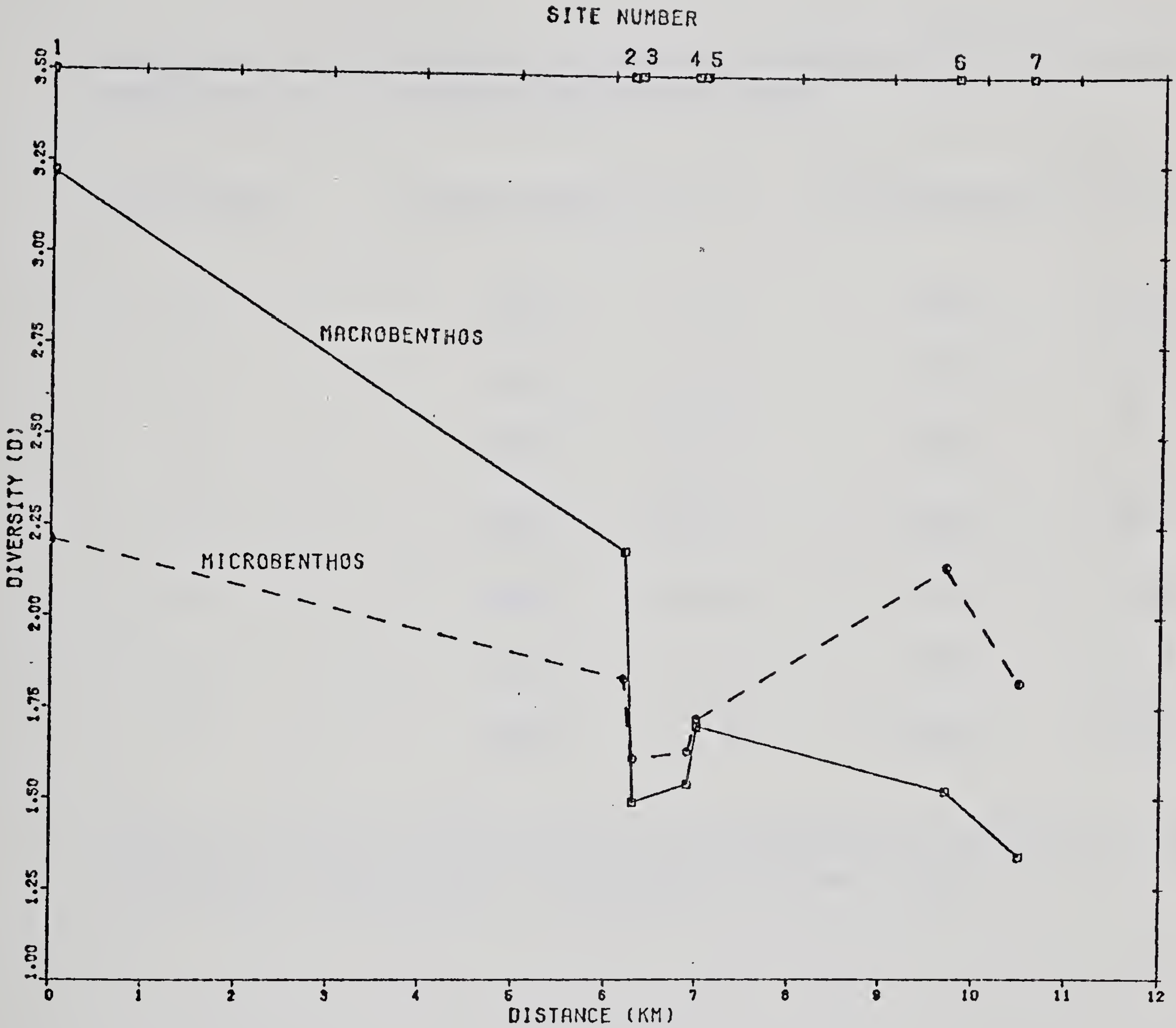


Figure 21. Comparison of macro- and microbenthic diversities. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

TABLE 6.

Densities of Tubificidae in the macrobenthos and microbenthos. Densities are in numbers per square meter.

Site number	Macrobenthos	%*	Microbenthos	%*
1	103	3.7	2650	96.3
2	364	5.7	5970	94.3
3	5760	21.5	21100	78.5
4	6590	39.7	10000	60.3
5	2860	23.5	9330	76.5
6	944	5.2	17300	94.8
7	1960	6.2	29900	93.8

* Percent of the total density of tubificids at that site. An average of all dates is presented for each site.

ratio of tubificid macrobenthic density to microbenthic density was higher at sites 3 to 5 than at the other sites. The relatively higher proportion of larger tubificids at the most disturbed sites suggests that there is either better survival of young animals or more rapid growth (so that individuals of the same age would be larger) at these sites.

As was true of macrobenthic diversity, microbenthic diversity showed much seasonal fluctuation, but with no distinct seasonal pattern (Fig. 22). Again, there appeared to be a general decline in diversity throughout the study period.

Potamoplankton

Potamoplankton (all invertebrates present in the water column, and collected in a 160 micron mesh plankton net) of Whitemud Creek was dominated by cyclopoid copepods and rotifers (Fig. 23 and Appendix 8). Both groups were abundant a short distance below the major storm sewers, the rotifers at site 4 and the copepods at sites 5 and 6. This suggests that these may be moderately pollution-tolerant organisms. The abundance of the animals at sites 4 and 5 perhaps indicates that recovery may be occurring more rapidly in the plankton than in the benthos.

Tubificids and chironomids were also abundant in the plankton (Fig. 23). Both were represented in the plankton almost exclusively by small individuals. Many were found entangled in bits of free-floating *Cladophora*. Of the remaining 20 taxa collected in the plankton samples, 5 were found mostly near the edge of the city, 7 were collected mainly near the mouth, and 8 were generally distributed (Appendix 8).

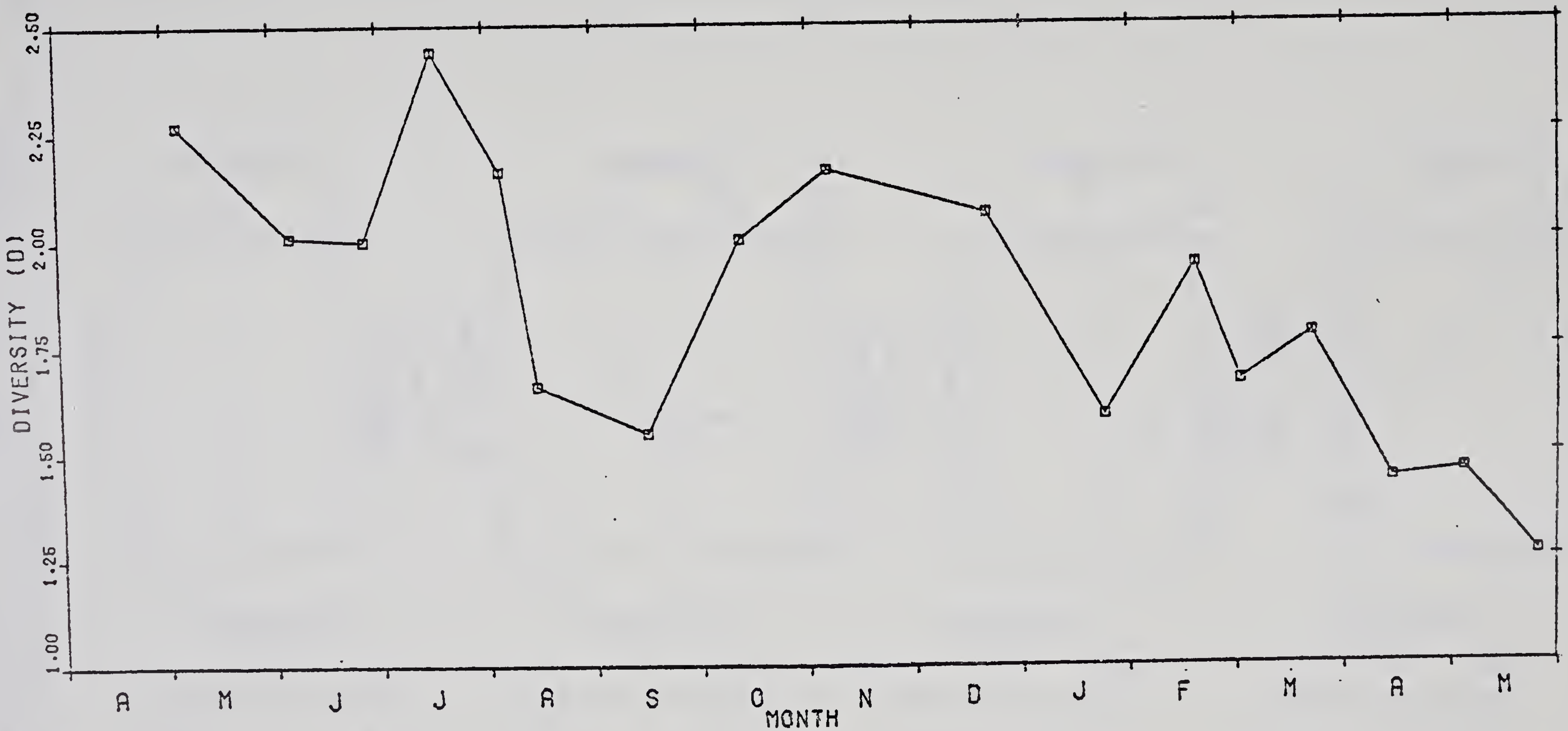


Figure 22. Seasonal variations in microbenthic diversity, 1976-1977. An average of all sites is represented for each date.

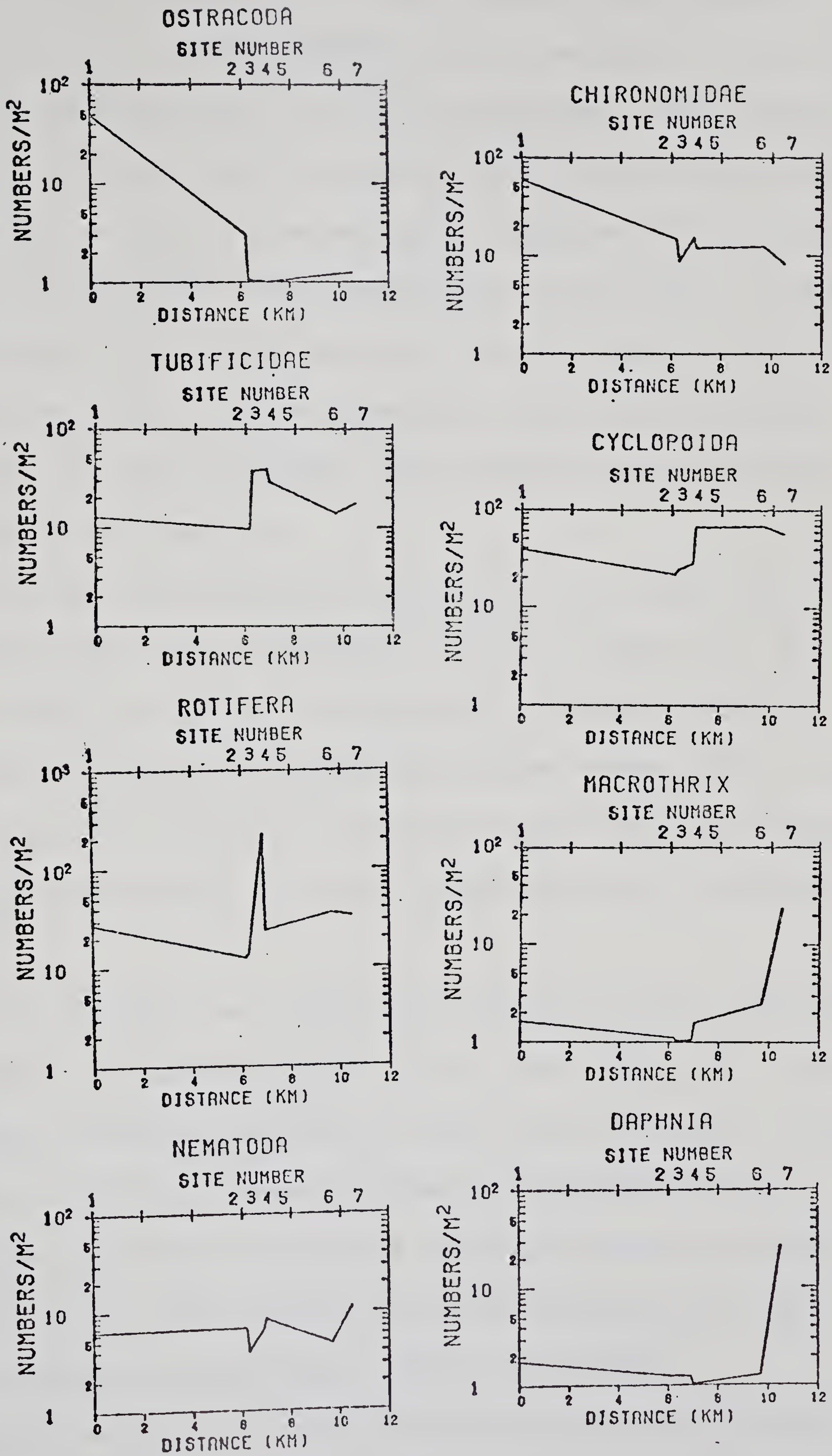


Figure 23. Spatial variations in numbers of some common potamoplankton. Distance is from edge of city moving downstream. An average of all dates ir presented for each site.

Those found primarily upstream of the city included early instars of benthic insects, such as hydropsychids, *Simulium* and *Baetis*, which were distributed in the same manner as in the macrobenthos. The planktonic individuals of all these taxa were probably ones that had been carried off the substrate as part of constant drift. Thus their distribution in the plankton should reflect their distribution in the benthos. However the distributions of many of the taxa that were more abundant in the plankton near the mouth, e.g. the cladocerans *Daphnia* and *Macrothrix*, the tardigrades and the collembolans, were probably due to the reduced current velocities at this site.

The mean richness of planktonic animals per site varied from a maximum of 16 at site 1 to a minimum of 6 at site 5 (Fig. 24). The pattern was similar to that of the macrobenthic richness. The main difference was the increase in planktonic richness between sites 4 and 5. This increase probably represents increasing recovery from the effects of discharge from storm sewer II, in spite of the minor input from storm sewer III.

Planktonic diversity and richness had similar patterns, with maximum values at site 1 and minimum values at sites 3 and 6 (Fig. 25). Planktonic and macrobenthic diversity patterns were also similar (Fig. 26). However the planktonic diversity increased instead of decreasing at site 7. This difference may reflect the reduced current velocities and greater water depth at site 7; this creates more lentic conditions and would favour true planktonic invertebrates, such as cladocerans.

Potamoplankton diversity showed no obvious patterns of seasonal variation (Fig. 27). However, as was true of the other sampling groups, there was a general trend of decreasing diversity throughout the course of the study.

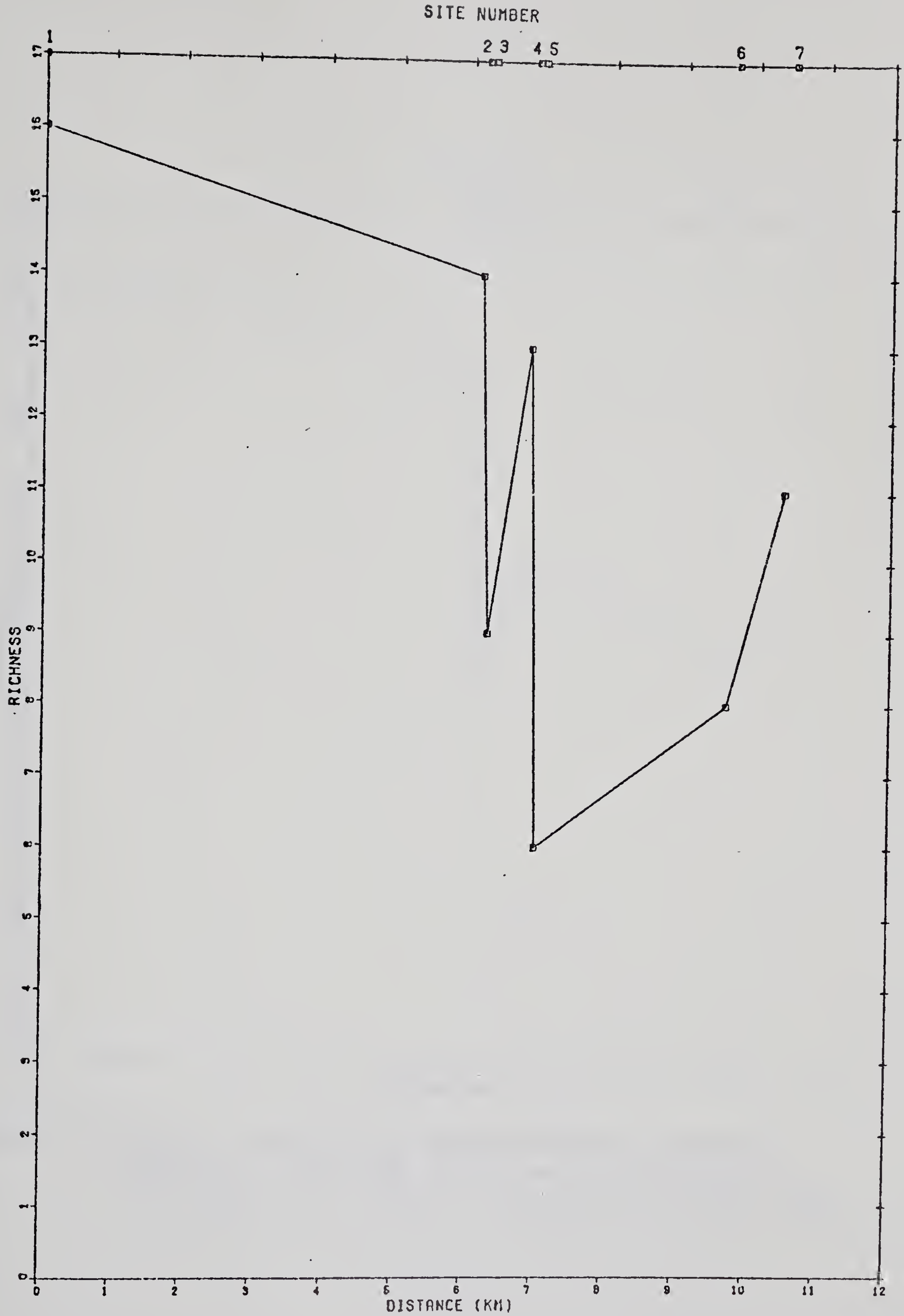


Figure 24. Spatial variations in potamoplanktonic richness. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

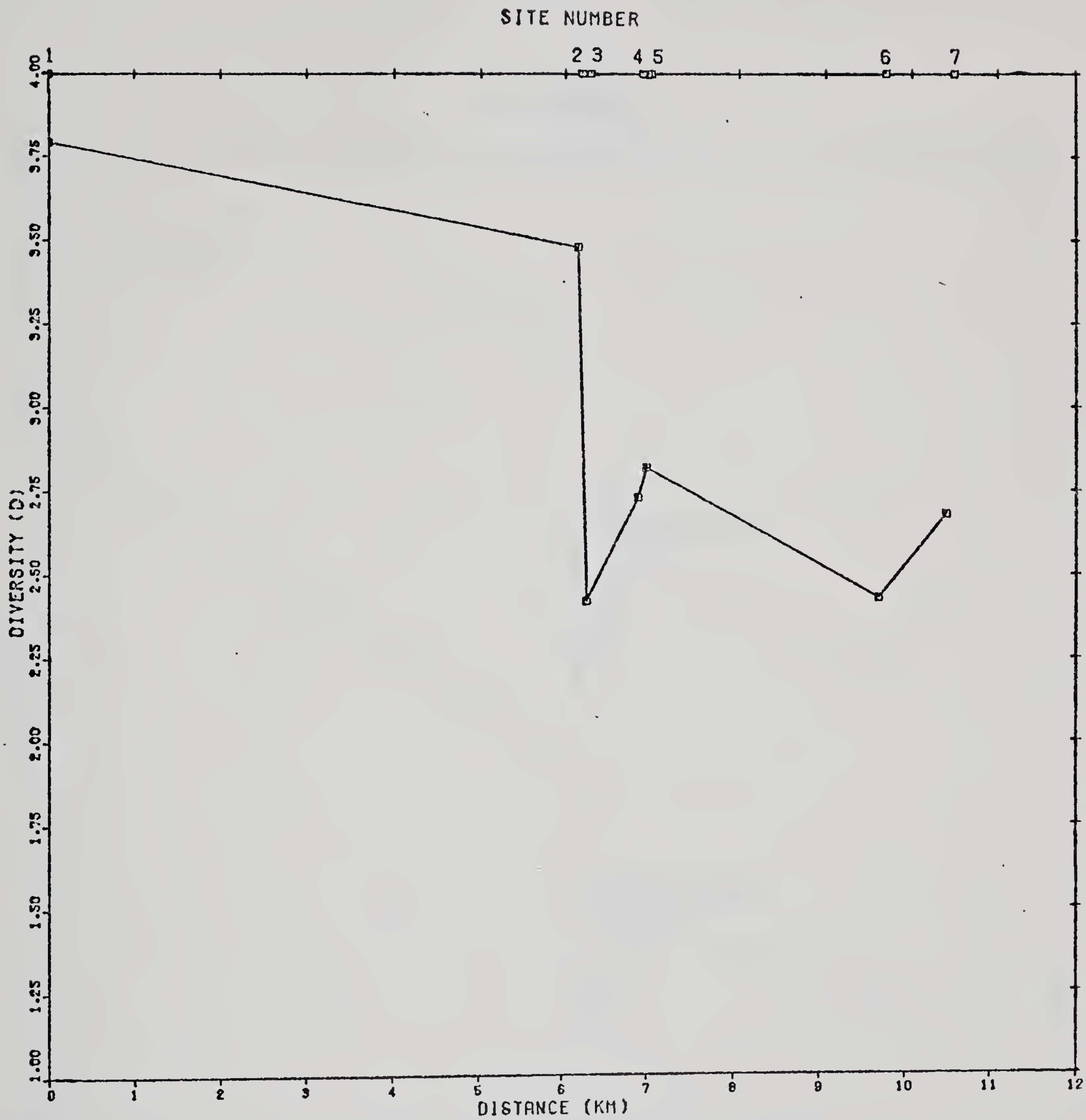


Figure 25. Spatial variations in potamoplanktonic diversity. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

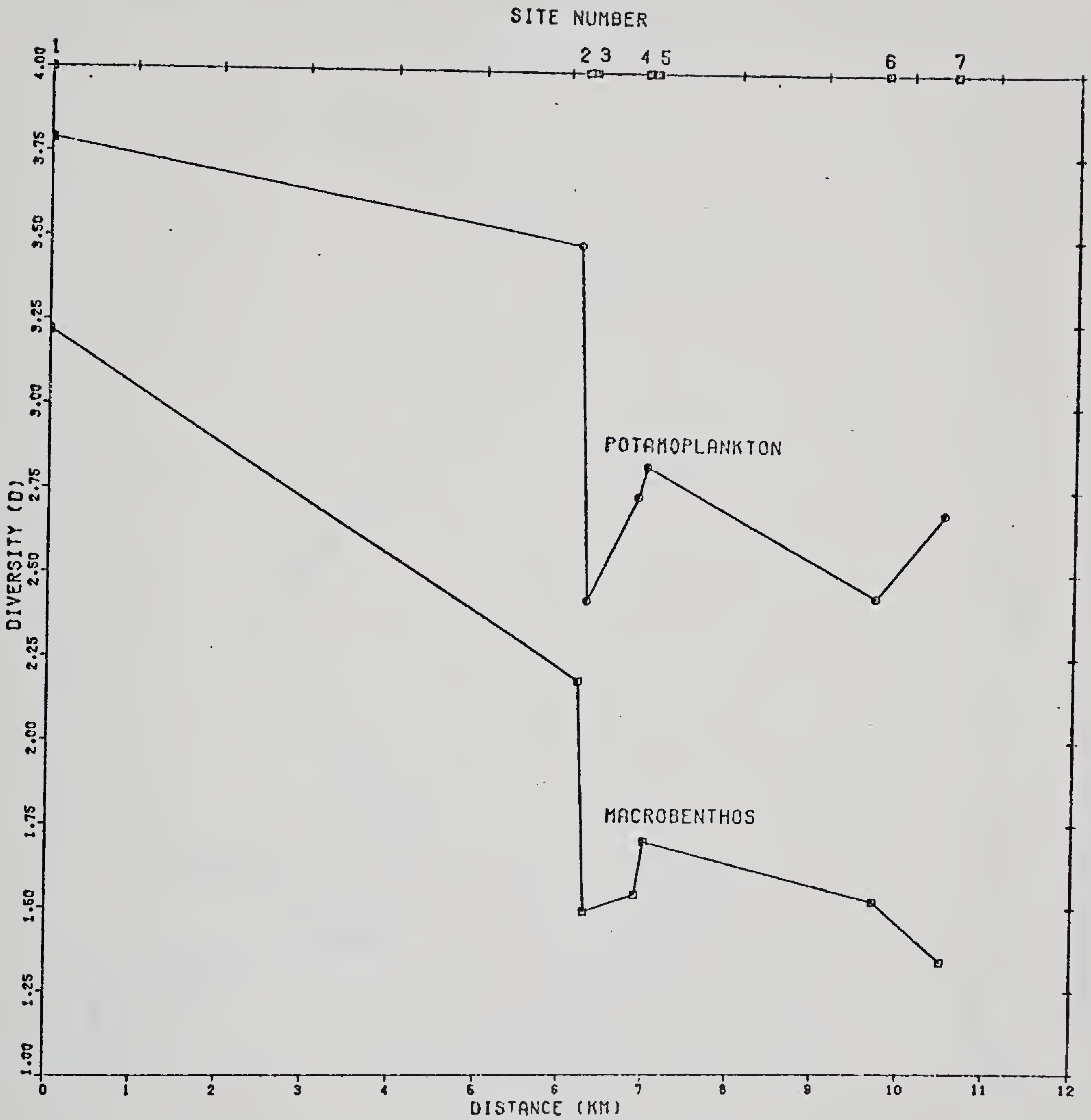


Figure 26. Comparison of macrobenthic and potamoplanktonic diversities. Distance is from edge of city moving downstream. An average of all dates is represented for each site.

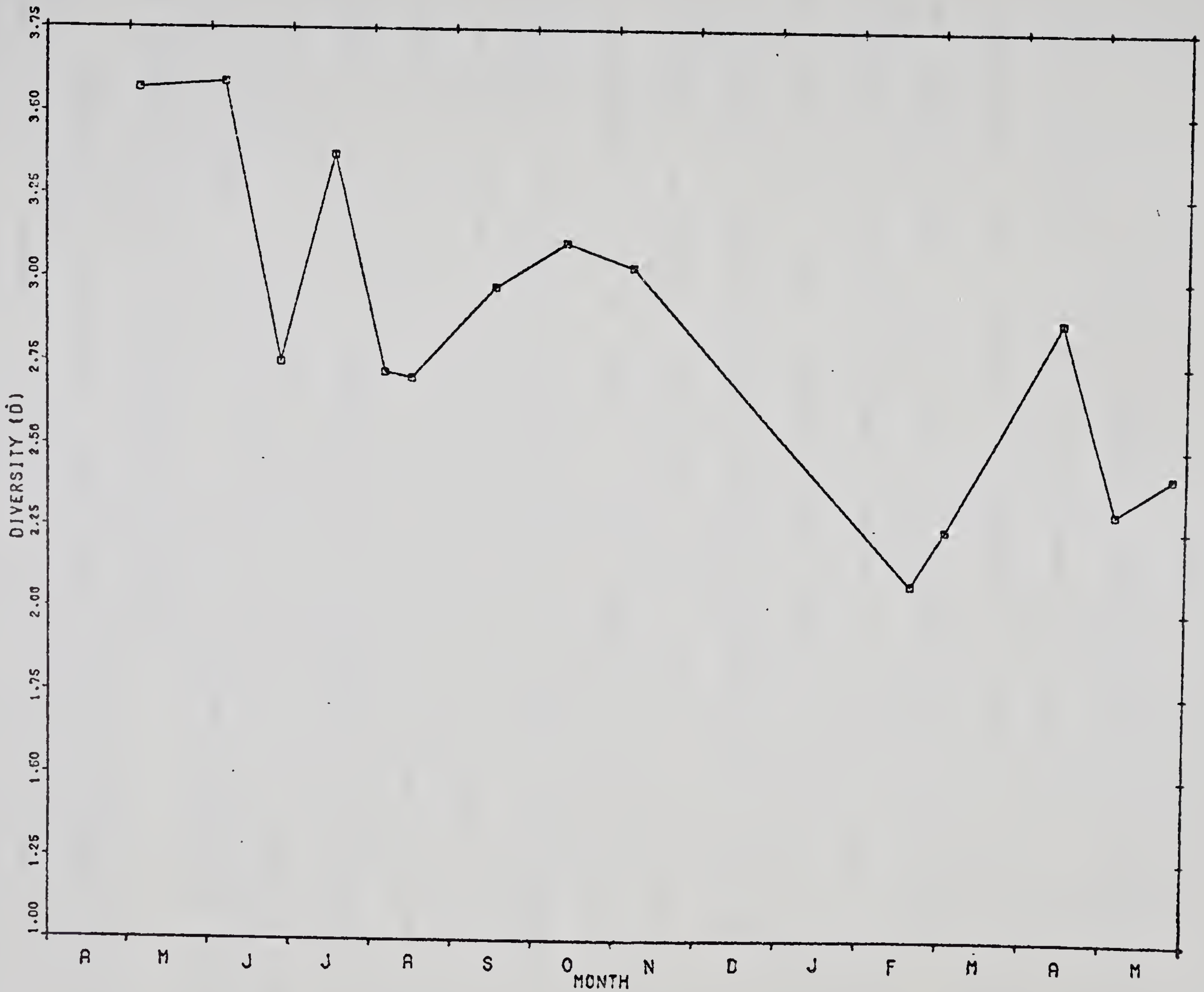


Figure 27. Seasonal variations in potamoplanktonic diversity, 1976-1977. An average of all sites is represented for each site.

Expected Runoff Effects

The effect of runoff from a storm sewer system at any point on a receiving stream should be proportional to the area drained by the system and inversely proportional to the distance from the point of discharge to the point where the effect is being experienced. The total effect at any site should be proportional to the sum of the effects of the individual storm sewer systems upstream from it. This "expected storm sewer runoff effect," in combination with the distance from the upstream edge of the city, can explain 98% of the spatial variation in macroinvertebrate diversity in Whitemud Creek. That is, a multiple linear regression of "expected storm sewer runoff effects" at each site, and distance from the upstream edge of the city to each site, on Simpson's diversity index produces a coefficient of determination (r^2) of .98 ($p < 0.01$). "Expected storm sewer runoff effects" at each site are calculated as $\Sigma\ (A/\sqrt{d})$ (the summation of the areas (A) drained by each storm sewer system upstream of the site divided by the square root of the distance (d) from the storm drain outlet to the site). Distances are in kilometers and Simpson's index is the mean over the study period at each site. The spatial variations in macrobenthic richness ($r^2 = .90$, $p < 0.05$) and density ($r^2 = .85$, $p < 0.05$) can also be explained in this way.

The partial correlation coefficients for the distance and "storm sewer effects" appear very similar, both for diversity and for richness. However the spatial variation in density is more strongly correlated with "storm sewer runoff effects" than with the distance from the upstream edge of the city.

The relationship with distance appears to represent a linear decrease in water quality (which would produce the observed variations in diversity). This in turn may be due to direct overland runoff (as opposed to storm sewer runoff). If the rate of direct runoff input is constant along the urban portion of the stream (whether or not there is a constant capacity for recovery), then the cumulative effect would increase linearly. If the relationship with distance from the edge of the city is due to direct runoff, then direct runoff, as well as storm sewer runoff, is important in producing the observed reductions in diversity and richness; but the increase in density is due mostly to storm sewer runoff (based on the partial correlation coefficients). Spatial variation in plankton diversity can also be explained in this manner (r^2 = .88, $p < 0.05$).

Seasonal variations in the fauna are more difficult to describe, because of the greater complexity of the factors influencing them. The amount of material entering Whitemud Creek via runoff depends upon the intensity and duration of rainfall incidents. Precipitation data are available from the Environment Canada Atmospheric Service. However data on the actual intensity of storms (rainfall rates) are not available. Rainfall intensity is important in determining the amount of material carried in the runoff. The lower the intensity of the precipitation, the greater the proportion of the moisture that will be absorbed by the substrate or evaporated, and hence will not be available to enter the stream as runoff. Also, the amount of material carried by runoff is not necessarily proportional to the total amount of runoff, regardless of intensity. After a sufficient

amount of runoff has occurred, many urban surfaces will have been swept clean, and subsequent runoff will carry no further material from these sources. In fact, the amount of material in storm sewer runoff has been observed to vary continuously during rainfall incidents (Cordery 1977). The amount of material in the runoff also depends on the amount that has accumulated on urban surfaces, which in turn depends on the amount of time since the last rains and on the percentage removal by these rains (Weibel et al. 1964, DeFilippi and Shih 1971, Sartor et al. 1974, Bradford 1977). A compounding factor is mechanical street cleaning operations that remove some, but by no means all, of the accumulated materials.

Runoff also results from snowmelt incidents. And it is difficult to estimate even the volume of runoff resulting from snowmelt, because of the lack of information about snowmelt rates during thaws and the moisture content of melting snow. Snowmelt rates are further influenced by road salting and sanding operations. Estimation of snowmelt is especially difficult during a winter such as that of 1976-1977, when there were numerous significant thaws throughout the winter, rather than just one major thaw in spring.

Also, as previously suggested, the life cycle patterns of the animals themselves may have important effects on observed seasonal variations in diversity and richness. Thus it is not possible to describe the seasonal variations in the fauna in terms of expected runoff.

GENERAL DISCUSSION

I found large changes in the invertebrate fauna of Whitemud Creek as it passes through the city of Edmonton, Alberta. There were reductions in diversity and numbers of taxa present and changes in the kinds of animals that dominate the fauna. Very similar spatial and temporal changes occurred in all three faunal components studied (the macrobenthos, microbenthos and potamoplankton).

According to the classification of Wilhm and Dorris (1968), the diversity indices calculated for the Whitemud Creek fauna would indicate that the stream downstream from the second storm sewer outlet was heavily polluted. The control site would be classified as moderately polluted (perhaps because of agricultural activity upstream from the city). Site 2 would also be moderately polluted, although diversity was occasionally low enough to indicate heavy pollution. However Wilhm and Dorris's classification was based on a species diversity index. And because an index based primarily on familial and generic groups was used in this study, Wilhm and Dorris's method would classify Whitemud Creek as being more polluted than it actually is.

Stream areas where tubificids comprise more than 80% of the fauna or are present in densities greater than $5000/m^2$ can also be considered to be heavily polluted by organic enrichment (Aston 1973). Tubificid dominance of 60% to 80% or densities of 1000 to $5000/m^2$, according to Aston (1973), indicate moderate pollution. By these standards, sites 3 and 4 were heavily polluted for most of the year, and when not heavily polluted, they were moderately polluted. Sites 5, 6 and 7 were moderately polluted.

[illegible]

[illegible]

The apparent causes of these changes were organic and inorganic enrichment, causing nocturnal oxygen deficits, and heavy siltation. The changes appeared to be the direct result of urban runoff, both from storm sewer discharge and from overland surface runoff. Unfortunately the measured chemical and physical constituents did not provide much information about actual conditions in the stream. This was likely because of the inability of standard techniques to measure total available nutrients. And of course chemical tests provide no information about past events in the stream.

The biological parameters studied indicated a general deterioration of conditions (re faunal diversity and richness) in the stream during the course of the study. This might have been due to the construction of the Whitemud Freeway Bridge, with its large scale surface disturbances. However the spatial variations in the fauna -- the recovery between sites 3 and 4 is evident in almost all measured biological parameters -- would indicate that this construction had a negligible effect. It is also possible that the housing developments north of 23 Avenue and west of 122 Street (contributing to overland runoff) and in Mill Woods (contributing to storm sewer runoff) might have been important in causing this deterioration. However if scarification of the immediate stream banks at the bridge construction site produced no noticeable effects, disturbances as far removed as Mill Woods should not either. Alternatively, the apparent deterioration of conditions might have been an artifact. One year's data are not enough to conclude that the stream is deteriorating.

There were potential inaccuracies in the determination of the biological parameters. The problem of animals lost due to mesh size has been reduced in this study by the use of both Hess and core samplers. While the 300 micron mesh used in the Hess sampler undoubtedly missed many of the smaller oligochaetes and chironomids, these animals were collected with the core sampler. Also, Mason, Lewis and Hudson (1975) have shown that missing some of the smaller individuals does not significantly affect calculated diversity index values. The use of diversity indices in this study and associated problems are outlined in Appendix 5. The most important errors in faunal assessment were probably due to the different sampling techniques that had to be used in winter. Techniques such as emergence trapping and continuous temperature monitoring could not be used in this study because of vandalism.

I assumed that observed variations in the invertebrate fauna of urban Whitemud Creek were due to urban runoff because of the strong relationship with expected urban runoff. And in the case of storm sewer runoff, there are no other potential causes for the observed patterns of variation. However other interpretations of the correlation between diversity and distance from the upstream edge of the city are possible. The observed faunal variations may have been due to natural longitudinal zonation within the stream. But I would not expect this much natural variation within such a short stretch of the stream. And the studies of Clayton (1975) and Visscher (1977) show no such changes along reaches of Whitemud Creek upstream from the city. Another possibility is that the variations were due to aerial inputs, resulting

from industrial air pollution. Aerial inputs would be expected to increase in intensity moving towards the center of the city. However, in a Missouri stream, Huff (1975, 1976) found that aerial inputs were insignificant relative to surface inputs, except for nitrates. While atmospheric pollution cannot be entirely ruled out as having affected spatial distribution in Whitemud Creek, I feel surface runoff was likely responsible for most of the variation.

CONCLUSIONS AND RECOMMENDATIONS

My study indicates a deterioration in the invertebrate fauna of Whitemud Creek as it flows through the city of Edmonton. Faunal diversity and richness were reduced. Tubificid worms increased in number to the point of dominating the fauna at some urban sites, and there were decreases in numbers or disappearance of many other groups of invertebrates. Alteration in the growth pattern of at least one animal was also demonstrated. These changes can be associated with, and appear to be the result of, urban runoff, including storm sewer discharge, and in particular the nutrient and silt load of this runoff.

Since the main cause of faunal deterioration in Whitemud Creek within the city appears to be urban runoff, the key to controlling the deterioration is to reduce the amount of material in the runoff entering the stream. Direct overland runoff can be controlled by maintaining vegetation along the stream to provide a filter for incoming runoff. Large scale surface disturbances in the vicinity of the stream should be avoided. Although one such large scale disturbance, the Whitemud Freeway Bridge construction, showed no obvious effects, this was probably due to the already poor water quality of the stream at this point. This kind of construction would likely have had noticeable effects at a less polluted site. Certainly other studies have shown significant physical, chemical and biological changes resulting from road and other construction activities (Guy and Ferguson 1962, White 1976, Barton 1977).

A number of things could be done to improve the quality of storm sewer discharge. Since much of the material in runoff arises from

paved surfaces, more effective street cleaning practices would be advantageous (Heaney and Sullivan 1971, Sartor et al. 1974). Current street cleaning is primarily for aesthetic reasons. These operations remove only large particle material and leave the smaller particles, which are responsible for most of the nutrient input (Sartor et al. 1974). Limitation of surface disturbances within storm drainage basins and better application of lawn fertilizers could also be expected to improve the quality of the runoff. However complete control of urban runoff water quality is impossible, and the best that can be hoped for is to minimize the effects on the receiving waters.

I believe that the major problem with water quality in Whitemud Creek is the quantity of incoming storm sewer runoff. I doubt that there will be any noticeable improvements unless the area being drained into Whitemud Creek by storm sewers is reduced.

The End

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APPENDIX 1. Temporal variations in physical and chemical constituents, 1976, 1977. Averages of all sites are presented.

	Total Phosphate (mg/1)	Ortho Phosphate (mg/1)	Meta and Poly Phosphate (mg/1)	Organic Phosphate (mg/1)	Total Nitrogen (mg/1)	Nitrate Nitrogen (mg/1)	Calcium Hardness* (mg/1)	Total Hardness* (mg/1)	Alkalinity* (mg/1)	Conductivity (micromhos)	pH	Dissolved Oxygen (% saturation)
May 2	2.04	1.22	0.61	0.20	2.05	0.06	127	189	178	502	8.36	89
June 5	1.50	0.71	0.48	0.32	2.08	0.25	172	274	198	612	7.81	105
June 26	1.37	1.22	0.11	0.06	0.87	0.51	122	185	139	448	7.65	88
July 17	1.27	0.80	0.29	0.20	1.59	0.72	214	338	191	752	7.78	91
Aug. 4	1.33	1.07	0.16	0.09	1.17	0.62	104	165	97	403	7.34	84
Aug. 25	1.16	0.62	0.29	0.24	1.42	0.41	203	343	206	864	7.71	94
Sept. 16	0.65	0.38	0.18	0.09	0.83	0.41	208	351	208	706	7.90	120
Oct. 11	0.56	0.35	0.12	0.08	0.62	0.22	235	417	232	744	7.77	87
Nov. 5	0.68	0.49	0.19	0.01	1.23	0.65	335	546	289	1150	7.92	76
Dec. 18	0.37	0.30	0.06	0.02	0.87	0.39	220	358	278	951	7.85	63
Jan. 24	0.63	0.46	0.12	0.05	1.52	0.54	347	533	334	1140	7.60	43
Feb. 17	1.18	1.11	0.03	0.04	1.49	0.36	229	371	249	1020	7.53	52
March 1	1.35	0.99	0.17	0.19	1.67	0.48	192	313	214	803	7.47	49
March 22	1.33	0.96	0.20	0.17	2.16	0.44	252	396	314	872	7.74	53
April 14	1.47	0.99	0.31	0.16	2.85	0.37	114	171	120	387	7.52	85
May 4	1.29	0.74	0.27	0.25	1.41	0.34	173	250	164	584	7.80	87
May 26	0.89	0.40	0.18	0.31	1.48	0.16	179	265	155	484	8.07	120

* as $CaCO_3$

APPENDIX 1. Continued.

	Sodium (mg/l)	Chloride (mg/l)	Chromium (mg/l)	Phenols (μg/l)	Color	Turbidity (JTU)	Algal Mat Growth (relative units)	Particulate Organic Matter (mg/l)	Depth of Silt Deposition (mm)	Air Temperature (^{o}C)	Water Temperature (^{o}C)	Flow (m^3/sec)
May 2	55	17.5	0.06	1.70	184	14.4	0.57	48	13.5	17.7	12.3	0.83
June 5	44	18.1	0.08	2.00	119	15.7	2.00	149	15.0	19.7	1.61	0.30
June 26	56	24.1	0.17	3.79	139	69.6	0.43	23	16.4	18.3	16.0	0.39
July 17	66	21.3	0.17	1.95	98	27.6	2.71	132	14.8	23.3	17.3	0.06
Aug. 4	34	10.6	0.13	2.86	114	61.0	1.43	61	13.0	22.6	18.6	0.59
Aug. 25	69	25.8	0.12	0.56	104	27.0	2.14	58	13.4	20.0	16.9	0.10
Sept. 16	74	19.1	0.12	4.91	65	9.1	2.14	43	17.0	18.3	14.0	0.15
Oct. 11	69	22.0	0.09	1.53	58	7.1	1.14	40	15.7	8.6	5.9	0.10
Nov. 5	77	22.4	0.08	4.17	74	5.5	3.00	122	15.6	5.2	2.0	0.07
Dec. 18	75	74.0	0.11	2.89	61	11.4	1.00	-	21.2	-1.7	0.6	0.06
Jan. 24	166	73.6	0.11	0.00	50	6.7	0.43	-	18.9	-3.9	0.1	0.02
Feb. 17	213	14.5	0.15	9.11	247	35.2	0.43	38	4.4	3.9	0.4	0.04
March 1	82	53.1	0.15	4.44	164	34.1	0.43	46	2.3	4.0	0.4	0.07
March 22	107	44.7	0.17	4.46	212	18.0	1.14	-	3.7	1.1	0.1	0.05
April 14	33	16.8	0.17	2.69	264	23.6	0.00	21	7.8	14.7	4.6	0.63
May 4	60	23.2	0.23	2.21	168	27.4	1.00	16	5.0	15.1	11.9	0.14
May 26	45	19.0	0.16	1.56	143	9.4	1.86	19	8.4	20.7	17.0	0.84

APPENDIX 2. Spatial variations in physical and chemical constituents. Averages of all dates are presented.

Site #	Total Phosphate (mg/1)	Ortho Phosphate (mg/1)	Meta and Poly Phosphate (mg/1)	Organic Phosphate (mg/1)	Total Nitrogen (mg/1)	Nitrate Nitrogen (mg/1)	Calcium Hardness* (mg/1)	Total Hardness* (mg/1)	Alkalinity* (mg/1)	Conductivity (micromhos)	pH	Dissolved Oxygen (% saturation)
1	0.97	0.53	0.26	0.18	1.30	0.10	157	247	314	719	7.87	73.8
2	0.82	0.50	0.22	0.11	1.21	0.26	212	334	217	740	7.89	83.3
3	1.12	0.78	0.23	0.12	1.72	0.61	268	417	231	878	7.84	81.1
4	1.45	1.03	0.24	0.19	1.70	0.55	211	324	181	756	7.71	71.1
5	1.48	1.08	0.26	0.13	1.85	0.61	213	349	187	778	7.69	82.0
6	1.18	0.84	0.18	0.17	1.69	0.38	190	322	172	721	7.55	82.4
7	0.85	0.55	0.15	0.14	0.95	0.26	141	226	153	457	7.68	85.6

* as $CaCO_3$

APPENDIX 2. Continued.

Site #	Sodium (mg/l)	Chloride (mg/l)	Chromium (mg/l)	Phenols (µg/l)	Color	Turbidity (JTU)	Algal Mat Growth (relative units)	Particulate Organic Matter (mg/l)	Depth of Silt Deposition (mm)	Air Temperature (^oC)	Water Temperature (^oC)	Flow (m^3/sec)
1	101	21.7	0.14	2.74	128	15.0	1.71	75.4	7.9	12.4	9.29	0.21
2	81	24.2	0.13	1.85	120	17.7	0.88	34.2	11.4	12.8	9.12	0.30
3	88	33.5	0.13	3.40	111	22.1	1.65	86.0	13.3	13.2	9.29	0.21
4	77	29.5	0.14	3.48	147	34.0	1.76	48.8	12.5	13.5	8.35	0.32
5	80	34.2	0.14	3.66	155	30.2	1.65	43.6	8.2	12.4	8.71	0.33
6	74	40.0	0.13	3.19	176	30.1	0.69	85.4	11.6	10.9	9.60	0.32
7	43	24.1	0.13	2.64	103	19.0	0.38	39.3	18.4	10.6	9.63	-

APPENDIX 3. Taxa found in Whitemud Creek. A = Abundant, C = Common, O = Occasional, R = Rare.

Cnidaria	*Hydra* sp.	O
Turbellaria		
Rhabdocoela		O
Nematoda		A
Rotifera		A
Tardigrada		O
Oligochaeta		
Tubificidae	*Limnodrilus hoffmeisteri*	A
	Limnodrilus profundicola	O
	Tubifex tubifex	A
Hirudinea		
Rhynchobdellida	*Glossiphonia complanata*	O
	Helobdella stagnalis	O
	Placobdella sp.	R
Pharyngobdellida	*Nephelopsis obscura*	O
	Erpobdella punctata	O
	Dina spp.	A
Crustacea		
Cladocera	*Daphnia* spp.	O
	Macrothrix spp.	O
	Chydorus spp.	O
Ostracoda		A
Copepoda: Cyclopoida		A
Amphipoda	*Hyallela azteca*	O
	Gammarus lacustris	C
Insecta		
Collembola		O
Ephemeroptera	*Tricorythodes stygiatus*	R
	Ephemerella inermis	R
	Caenis forcipata	A
	Leptophlebia sp.	O
	Heptagenia hebe	C
	Baetis sp.	C

APPENDIX 3. Continued.

Insecta			
Odonata		*Enallagma* spp.	R
		Ischnura spp.	R
		Aeshna spp.	R
		Libellula spp.	R
Plecoptera		*Brachyptera* sp.	R
		Zapada sp.	R
		Capnia sp.	R
		Isoperla sp.	R
Hemiptera:	Corixidae		A
	Gerridae		C
Coleoptera:	Haliplidae	*Haliplus* sp.	O
	Dytiscidae	*Hydroporus* sp.	C
		Ilybius sp.	O
	Elmidae		O
	Hydrophilidae		R
	Staphylinidae		R
	Amphizoidae		R
Trichoptera:	Hydropsychidae	*Hydropsyche* spp.	C
		Cheumatopsyche spp.	C
	Hydroptilidae		R
	Phryganeidae	*Ptilostomis* sp.	R
	Limnephilidae		R
	Leptoceridae	*Oecetis* sp.	R
Diptera:	Tipulidae	*Tipula* sp.	R
		Pedicia spp.	O
		Hexatoma spp.	O
	Psychodidae	*Pericoma* sp.	O
		Psychoda sp.	O
	Heleidae		O
	Chironomidae:	Tanypodinae	A
		Orthocladinae	A
		Tanytarsini	A
		Chironomini	A
	Simuliidae	*Simulium* spp.	A
	Dixidae	*Dixa* sp.	R
	Stratiomyidae		R
	Tabanidae	*Tabanus* spp.	C
	Rhagionidae	*Atherix* sp.	R
	Dolichopodidae		O
	Empididae		R
	Ephydridae		R
Arachnida			
Acari			O

APPENDIX 3. Continued.

Mollusca		
Gastropoda	*Physa* spp.	C
	Lymnaea spp.	O
	Gyraulus spp.	O
	Helisoma sp.	R
	Ferrissia rivularis	O
Pelecypoda	*Anodonta grandis*	R
	Sphaerium spp.	C
	Pisidium spp.	C

APPENDIX 4. Spatial variations in densities (numbers/m^2) of macroinvertebrates. Averages of all dates are presented.

	Site #						
	1	2	3	4	5	6	7
Nematoda	7.10	5.11	3.03	3.47	1.74	0.48	10.8
Tubificidae	103	364	5760	6590	2860	943	1970
Glossiphonia	0.89	1.78	0.0	0.77	0.0	6.48	0.72
Helobdella	0.95	0.93	0.0	0.0	0.0	0.24	0.72
Erpobdella	0.67	0.89	0.19	0.0	0.0	0.48	0.24
Dina	2.91	9.77	1.71	5.21	7.71	18.3	2.88
Nephelopsis	0.0	0.87	0.35	0.17	0.0	0.0	0.17
Hyallela	4.91	0.22	0.0	0.0	0.0	0.0	0.0
Gammarus	17.5	0.0	0.0	0.0	0.0	0.0	0.0
Caenis	166	55.3	12.1	16.2	19.7	5.28	4.32
Heptagenia	6.22	0.22	0.0	0.19	0.19	0.0	0.0
Leptophlebia	0.84	0.22	0.0	0.0	0.0	0.12	0.24
Baetis	10.7	17.1	12.0	0.77	0.19	0.0	0.0
Corixidae	3.04	5.11	0.57	0.96	0.58	0.48	290
Hydropsyche	42.6	22.6	2.66	2.31	5.01	0.0	0.0
Cheumatopsyche	83.0	13.6	0.76	0.77	0.39	0.24	0.0
Haliplus	0.89	0.44	0.0	0.0	0.0	0.0	0.0
Hydroporus	1.18	5.77	0.0	0.58	0.19	5.52	4.68
Ilybius	0.0	0.0	0.0	0.17	0.17	0.0	0.0
Elmidae	0.71	0.67	0.0	0.19	0.0	0.0	1.22
Hexatoma	0.89	0.0	0.0	0.0	0.0	0.0	0.0
Pedicia	1.00	0.02	0.0	0.0	0.0	0.0	0.0
Psychoda	0.0	0.0	1.52	0.0	0.0	0.0	0.0
Pericoma	0.40	0.0	0.19	0.0	0.0	0.0	0.0
Simulium	307	146	9.30	1.35	0.19	0.0	0.0
Chironomidae	190	177	5593	250	89.6	133	59.9
Heleidae	1.55	0.44	0.0	0.39	0.19	0.05	1.20
Tabanus	5.77	0.22	0.40	0.19	0.19	0.24	0.0
Dolichopodidae	0.67	0.22	0.0	0.19	0.0	0.24	0.0
Physa	5.62	2.00	0.95	2.57	0.96	0.72	7.68
Lymnaea	1.11	0.04	0.19	0.0	0.0	0.0	0.38
Ferrissia	0.22	0.22	0.0	0.60	2.76	0.02	0.0
Gyraulus	0.17	0.17	0.17	0.36	0.0	0.35	1.04
Sphaerium	5.35	0.44	0.0	1.19	0.0	0.24	3.67
Pisidium	13.8	2.71	0.0	3.20	0.41	0.72	3.79

APPENDIX 5. Calculation of Simpson's diversity index.

The diversity index used in this study was calculated according to Simpson's formula (Simpson 1949):

$$D = 1/\Sigma p_i^2$$

where "D" is the diversity of the sample in dimensionless units and "p_i" is the proportion of the i^{th} taxon in terms of total numbers. The sum is inverted so that "D" will increase as diversity increases. The minimum possible diversity, when all individuals in the sample belong to the same taxon, is 1. The maximum possible is equivalent to the number of taxa in the sample.

Because it was not possible to identify many of the animals to species, or in some cases even to genus, the diversity index was calculated on the basis of recognizable taxa. Thus a mixed diversity index, involving primarily generic and familial groups, rather than a species diversity index, was calculated. Other workers have shown that higher taxonomic groups, such as family or even class, can be used for measuring community diversity (Pielou 1967, Egloff and Brakel 1973). However the higher the taxonomic level used, the lower the sensitivity of the index to environmental perturbations (Egloff and Brakel 1973). Thus I decided to use a mixed diversity index to obtain the maximum sensitivity from the data gathered. Although it is more sensitive, the pattern of change in the mixed diversity index was very similar to that of an index based on familial groups alone, as shown by the high correlation between the two ($r = .98$, $p<.001$).

Simpson's index was chosen over the more commonly used Shannon-Wiener index because of the former's tendency to de-emphasize

rare taxa (Peet 1974). There is potential for movement of a few individuals from the upstream reaches of Whitemud Creek or from the North Saskatchewan River into regions where they do not normally occur. Examples are the occasional presence of a few stonefly and *Ephemerella* nymphs at site 7. Thus the use of a diversity index that emphasizes rare taxa, such as Shannon-Weiner's, would not provide an accurate assessment of environmental conditions under such circumstances. Many workers suggest that Simpson's index is a satisfactory, if not preferred, index (Hiep and Engels 1974, Peet 1974).

APPENDIX 6. Spatial variations in biomass (g/m^2) of macroinvertebrates. Averages of all dates are presented.

	Site #							
	1	2	3	4	5	6	7	%*
Nematoda	.014	.018	.008	.013	.002	0	0	0.27
Tubificidae	.323	.765	3.250	8.600	2.090	2.260	5.910	48.72
Glossiphonia	.037	.032	0	.011	0	.042	.065	0.37
Helobdella	.026	.016	0	0	0	.007	.018	0.12
Erpobdella	.331	.469	.059	0	0	.037	.225	1.08
Dina	.201	.755	.342	1.160	.470	1.700	.725	8.32
Nephelopsis	0	.337	.099	.039	0	0	.012	0.70
Hyallela	.069	.010	0	0	0	0	0	0.18
Gammarus	.681	0	0	0	0	0	0	1.29
Caenis	.266	.284	.044	.126	.057	.077	.033	3.08
Heptagenia	.047	.005	0	.006	.004	0	0	0.17
Leptophlebia	.020	.002	0	0	0	0	.008	0.17
Baetis	.046	.103	.072	.021	0	0	0	0.75
Corixidae	.165	.065	.010	.038	.014	.024	1.700	2.15
Hydropsyche	.794	.426	.042	.035	0.60	0	0	2.22
Cheumatopsyche	1.000	.022	.003	.003	.005	.004	0	1.77
Haliplus	.012	.081	0	0	0	0	0	0.08
Hydroporus	.047	.124	0	.024	.064	.139	.195	1.35
Ilybius	0	0	0	.003	.022	0	0	0.21
Elmidae	.014	.051	0	.086	0	.035	.030	0.27
Hexatoma	.020	0	0	.041	0	0	0	0.25
Pedicia	.003	0	0	0	0	0	0	0.08
Psychoda	.058	0	.013	0	0	0	0	0.10
Pericoma	0	0	0	0	.005	0	0	0.02
Simulium	.522	.246	.072	.004	.005	0	0	2.89
Chironomidae	.397	.335	.447	.496	.245	.302	.331	14.08
Heleidae	.029	.004	0	.007	.012	0	.020	0.38
Tabanus	.443	.005	.031	.018	0	.006	0	0.83
Dolichopodidae	.004	.004	0	.016	0	.010	0	0.27
Physa	.864	1.450	.402	.781	.044	.024	1.960	3.30
Lymnaea	.071	.111	.008	0	0	0	.019	0.22
Ferrissia	0	0	0	.044	.081	0	0	1.30
Gyraulus	.012	.006	.009	.013	0	.160	.050	0.49
Sphaerium	.201	.008	0	.019	0	.144	.203	0.90
Pisidium	.546	.047	0	.187	.027	.023	.095	2.07

* average % of total macroinvertebrate fauna over all sites.

APPENDIX 7. Spatial variations in densities (numbers/m^2) of microinvertebrates. Averages of all dates are presented.

	Site #						
	1	2	3	4	5	6	7
Rhabdocoela	182	144	96	260	91	0	0
Rotifera	2090	4010	2660	885	3550	18800	4000
Nematoda	8790	11900	15900	4580	16000	14400	12000
Tubificidae	2650	5960	21100	9960	9320	17200	29900
Cyclopoida	3300	201	150	872	1090	7950	5190
Ostracoda	5300	359	0	91	0	324	93
Caenis	30	101	14	13	0	0	0
Baetis	15	114	0	0	0	0	0
Corixidae	15	0	1200	13	0	0	1160
Hydropsychidae	742	0	0	0	0	0	0
Hydroporus	0	72	0	0	0	0	0
Simulium	1370	2280	430	52	0	0	0
Chironomidae	12000	6180	7750	2300	1290	17300	1500
Ferrissia	0	0	0	26	0	0	0
Sphaerium	60	14	0	0	0	15	463
Pisidium	74	72	0	13	0	0	108

APPENDIX 8. Spatial variations in densities (numbers/m^2) of potamoplankton. Averages of all dates are presented.

	Site #						
	1	2	3	4	5	6	7
Hydra	1.96	0.04	0.03	0.00	0.03	1.68	0.39
Rhabdocoela	1.09	0.39	0.36	0.49	0.15	0.50	0.51
Rotifera	26.8	11.6	12.8	223	23.0	34.5	32.3
Nematoda	5.43	5.85	2.97	5.70	7.52	3.78	10.0
Tardigrada	0.00	0.11	0.00	0.00	0.00	0.08	0.50
Tubificidae	11.8	8.65	36.4	38.2	28.4	12.9	16.5
Macrothrix	0.63	0.11	0.03	0.06	0.58	1.45	22.1
Daphnia	0.77	0.25	0.26	0.24	0.03	0.27	25.2
Chydorus	1.68	1.23	0.87	0.28	0.24	0.61	0.12
Cyclopoida	37.8	20.9	23.3	27.5	65.5	67.6	56.8
Ostracoda	48.7	2.07	0.06	0.03	0.00	0.19	0.23
Collembola	0.04	0.11	0.19	0.40	0.31	0.34	0.08
Caenis	2.80	0.53	0.48	0.70	0.95	0.34	0.04
Heptagenia	0.07	0.04	0.00	0.00	0.06	0.00	0.00
Baetis	0.46	0.49	0.55	0.03	0.03	0.15	0.04
Corixidae	0.70	0.25	0.39	0.33	0.06	0.23	4.18
Hydropsychidae	0.35	0.04	0.00	0.00	0.00	0.04	0.00
Hydroporus	0.21	0.18	0.00	0.00	0.03	0.08	0.59
Simulium	9.24	2.21	0.16	0.03	0.21	0.11	0.00
Chironomidae	58.2	13.7	7.82	14.2	11.1	11.4	7.31
Acarina	0.14	0.04	0.06	0.03	0.06	0.00	0.47

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